Nutrient transport, transformation, and retention in urban landscapes

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Dissertation abstract

Urban nutrient sustainability faces challenges of both too much and too little: Excess nutrient loading to the environment can degrade ecosystem functions and impact human health, while at the same time depleting nonrenewable nutrient sources and moving nutrients into unrecoverable pools. Most studies and efforts to date have focused on source reduction, identifying and reducing the largest drivers of carbon (C), nitrogen (N), and phosphorus (P) consumption. However, this addresses only one aspect of urban nutrient cycling; processes that transport, transform, or retain nutrients also determine their eventual fate as pollution, inert storage, or recycling.

The first chapter examined C, N, and P output fluxes from ~2,700 households in the Twin Cities metropolitan area (Minneapolis-Saint Paul, Minnesota, USA), and tracked these fluxes through various transformations in the waste streams to their eventual fates. We found few opportunities to redirect pollutant fluxes to either inert storage or recycling; reducing household nutrient pollution must rely primarily on reducing consumption. High pollution fluxes were driven not only by household nutrient outputs, but also by waste-management practices (e.g. septic vs. sewer) and spatial considerations. In contrast, we found substantial opportunities to increase household N and P recycling by ten-fold, which could potentially exceed household inputs of N and P in food.

To complement this study of opportunities for improving nutrient waste management, the second and third chapters examined opportunities to manage the biophysical environment – specifically, the urban forest – to reduce nutrient pollution. We focused on the role of urban trees driving N and P movement from land to water, both leaching to groundwater and loading to stormwater. In the second chapter, we compared nutrient leaching under 33 trees of 14 species, as well as open turfgrass areas, and explored correlations with soil nutrient pools and plant functional traits. Trees had similar or lower N leaching than turfgrass in 2012 but higher N leaching in 2013; trees reduced P leaching compared with turfgrass in both 2012 and 2013, deciduous trees more than evergreens. Scaling up our measurements to the Capitol Region Watershed (~17,400 ha), we estimated that trees reduced P leaching to groundwater by 533 kg in 2012 and 1201 kg in 2013.

Removing the same amounts of P with stormwater infrastructure would cost $2.2 million and $5.0 million per year, respectively.

In the third chapter, we measured tree litter nutrient inputs to street gutters, which can ultimately contribute to stormwater loading, under four species of boulevard trees. Differences among tree species in the total amount of nutrients in the street gutters were driven primarily by interspecific differences in the mass of litter dropped, which were much greater than differences in litter chemistry. In developing management recommendations, we found that tree phenology is a more important consideration than litter chemistry. Cleaning up spring and autumn pulses of tree litter shortly after they fall has substantial potential to reduce nutrient inputs to stormwater; for autumn litterfall, we estimated that doing so could remove 219.0-274.4 kg N km^{-2} and 14.2-20.6 kg P km^{-2}. Because of the wide variation in species' litterfall timing, achieving this goal is likely to require adjusting both boulevard tree selection and litter cleanup strategies.

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Introduction

Urban nutrient sustainability faces challenges of both too much and too little: Excess nutrient loading to the environment can degrade ecosystem functions and impact human health, while at the same time depleting nonrenewable nutrient sources by moving nutrients into unrecoverable pools. Carbon (C), nitrogen (N), and phosphorus (P) are essential elements, but too much in certain forms can contribute to global climate change, local and regional air pollution, terrestrial N deposition, and water pollution. At the same time, society also faces shortages of essential elements leading to nutrient scarcity. Most urban C and P fluxes come from nonrenewable sources – respectively, fossil fuels and phosphate rock – and converting N to useable fertilizer requires large inputs of energy, currently from fossil fuels. As the human population grows and shifts increasingly to urban and peri-urban areas, understanding and improving urban nutrient cycling is an essential part of working to mitigate nutrient pollution and nutrient scarcity.

Most studies and efforts to date have focused on source reduction, identifying and reducing the largest drivers of C, N, and P consumption. However, this addresses only one aspect of urban nutrient cycling; processes that transport, transform, or retain nutrients also determine whether nutrients end up as pollution, inert storage, or recycling. Understanding and improving these processes is essential for identifying and working with households that contribute the largest amounts of nutrient pollution, improving waste-management practices to increase nutrient recycling, and designing and managing urban vegetation to retain nutrients on land instead of allowing them to enter and pollute local lakes and streams.

In this dissertation, I examine C, N, and P transport, transformation, and retention in the Twin Cities metropolitan area (Minneapolis-Saint Paul, Minnesota, USA). Chapter 1, part of the Twin Cities Household Ecosystem Project (TCHEP), is a broad overview of nutrient output fluxes from ~2,700 households along an urban to peri-urban gradient. In it, we quantified the eventual fates of household nutrient fluxes by tracing C, N, and P outputs through various transformations in the waste streams to determine the amount and forms that ended up as nutrient pollution, inert storage, or nutrient recycling. We identified

pollutant fluxes where a small proportion of households emitted a large proportion of the pollutant; source-reduction efforts are most likely to succeed if they work specifically with this "high-emitter" subset of households. We also quantified opportunities for improving waste management to substantially increase N and P recycling to the food-production system.

Narrowing the scope from this broad overview, Chapters 2 and 3 focus on one specific aspect of urban nutrient cycling: the role of urban trees driving nutrient movement from land to water. Chapter 2 explores the extent to which urban trees retain N and P leaching through the soil to groundwater, and Chapter 3 quantifies litter inputs from boulevard trees to street gutters, which potentially increases N and P transport to stormwater. Both stormwater and groundwater are important pathways for N and P entering local lakes and streams. Excess nutrients feed algae blooms, resulting in bad odor and taste, decreased water clarity and dissolved oxygen, and the loss of desirable species. In both chapters, we compare different common species of urban trees and examine relationships between tree physiological traits, especially litter chemistry, and nutrient transport or retention. In Chapter 3 we also explore how the timing of litter inputs, which can vary considerably among species, interacts with the timing of street sweeping to determine whether litter in the street gutter gets swept up or washed down the storm drains. These results can inform urban forest planning and management to help protect water quality by reducing N and P entering local lakes and streams.

Chapter 1:

Reduce, redirect, or recycle? Quantifying opportunities to increase household nutrient sustainability.

Abstract

Excess nutrient loading to the environment can degrade ecosystem functions and impact human health (nutrient pollution), while at the same time depleting nonrenewable nutrient sources by moving nutrients into unrecoverable pools (nutrient scarcity). In this study, part of the Twin Cities Household Ecosystem Project (TCHEP), we examined carbon (C), nitrogen (N), and phosphorus (P) output fluxes from ~2,700 residential households in the Minneapolis-Saint Paul metropolitan area (Minnesota, USA). We quantified the eventual fates of household nutrient fluxes by tracing C, N, and P outputs through various transformations in the waste streams. We identified specific fluxes that may be a small proportion of total fluxes but cause significant environmental or health impacts, in order to guide focused efforts to reduce these specific fluxes. We also identified and quantified opportunities for redirecting pollutant fluxes to non-polluting forms or increasing recycling of C, N, and P.

We found few opportunities to redirect pollutant fluxes to non-polluting forms; reducing household nutrient pollution must rely primarily on reducing consumption. High pollution fluxes were driven not only by household nutrient outputs, but also by waste-management practices (e.g. septic vs. sewer) and spatial considerations. For example, our data suggest that reducing potential human exposure to nitrogen oxides or wood smoke pollutants can be best achieved by focusing not on the highest emitters, but rather on households in densely populated neighborhoods. In contrast, we found substantial opportunities to increase household N and P recycling by ten-fold, which could potentially return more N and P to the food-production system than households consume in food. Doing so would require the recycling of nearly all food waste and yard waste (to livestock, cropland, or gardens), implementation of industrial-scale P recovery from incinerator ash, and separation of urine

for fertilizer use. Our results suggest that improving waste management may be an important, though not sole, means to increase nutrient recycling and mitigate nutrient scarcity.

Introduction

Urban nutrient sustainability faces challenges of both too much and too little: Excess nutrient loading to the environment can degrade ecosystem functions and impact human health (nutrient pollution), while at the same time depleting nonrenewable nutrient sources by moving nutrients into nonrecoverable pools (nutrient scarcity). In this study, part of the Twin Cities Household Ecosystem Project (Fissore et al. 2011), we addressed urban nutrient sustainability by examining carbon (C), nitrogen (N), and phosphorus (P) output fluxes from residential households and tracking these nutrient outputs through transformations in waste streams to their eventual fates as pollutants, inert forms, or recycled nutrients. We quantified opportunities for changes in nutrient management strategies to reduce nutrient pollution and increase nutrient recycling. By examining C, N, and P fluxes simultaneously, we also considered synergies and tradeoffs among the three elements when evaluating management options.

In order to meet the twin challenges of nutrient pollution and nutrient scarcity, sustainability efforts must address the substantial C, N, and P fluxes in urban and suburban areas. Cities and suburbs make up only 4-6% of land area in the United States (McPhearson et al. 2013) but contain 80% of the U.S. population, with urban population growth outpacing rural growth (US Census Bureau 2012). Globally, over half of the world's population now lives in urban areas, projected to reach 60% (5 billion people) by 2030 (United Nations Population Fund 2007). Nutrient fluxes are not only high and concentrated in urban areas, but also more spatially variable than in undeveloped or agricultural systems and controlled by a number of unique input, transport, and transformation processes (Kaye et al. 2006, Grimm et al. 2008). Urban ecosystem ecology has made significant advances in the last decade to understand unique aspects of urban nutrient cycling, but the current state of knowledge is not sufficient to guide transitions to sustainable cities (Pickett et al. 2011).

In this study, as throughout TCHEP, we examine urban nutrient cycling not at the broad scale of sector-level analyses or whole-city nutrient budgets, but at the scale of household-level nutrient fluxes. Household fluxes are governed by many individuals' choices and decisions, and households in the same urban area can have very different nutrient fluxes from one another. We focus on characterizing the similarities and differences among our study households, especially evaluating the disproportionality among household fluxes: namely, the degree to which a small number of households contribute a large proportion of total household nutrient fluxes. This makes it possible to design efforts that help households reduce nutrient consumption, increase nutrient recycling, or take other steps to increase nutrient sustainability. Fluxes with low disproportionality (relatively even distributions) are likely to be best addressed through efforts that work with all households, since all households are generating relatively similar fluxes. In contrast, fluxes with high disproportionality (skewed distributions) may be better addressed by identifying and focusing on the subset of high-flux households, since a small proportion of households are generating a large proportion of the total nutrient fluxes. TCHEP also couples these biophysical assessments with analyses of household characteristics (e.g. physical house attributes, demographics) and behaviors (e.g. values, attitudes, knowledge, norms) to understand what drives household nutrient fluxes and how to help increase nutrient sustainability.

We define nutrient sustainability not as an ideal end state, but rather as a directional improvement that reduces nutrient pollution and the use of nonrenewable nutrient sources. (Increasing the social and economic benefits of nutrient use is also an important goal of nutrient sustainability, but assessing this would require a more comprehensive analysis of social and economic factors than is within the scope of our study.) Understanding the fate of nutrient fluxes through households – the goal of this study – is important in working to increase urban nutrient sustainability. First, excess nutrients in certain forms and in certain pools can impair ecosystem functioning and/or human health. For example, household C, N, and P fluxes contribute to global climate change, local and regional air pollution, terrestrial N deposition, and water pollution. Second, at the same time that excess nutrients are causing pollution, household nutrient use is depleting nonrenewable nutrient sources and

moving these nutrients into nonrecoverable pools. Much of the C and P in household nutrient fluxes comes from nonrenewable sources (fossil fuels and mined phosphate rock, respectively), and much of the N requires fossil-fuel energy to convert it from atmospheric N_2 into usable fertilizer.

Too Much: Nutrient pollution

C, N, and P are essential elements, but too much in certain places and forms can impair ecosystem functioning and/or human health by contributing to global climate change, local and regional air pollution, terrestrial N deposition, and water pollution. We quantified the fates of household nutrient fluxes to determine the extent to which households contribute to these four types of nutrient pollution.

Global climate change: Household combustion of fossil fuels (e.g. air and car travel, household energy use) contribute to global climate change by emitting carbon dioxide (CO_2), methane (CH_4), and nitrous oxide (N_2O). Household nutrient fluxes also include precursor compounds that contribute to greenhouse-gas formation, such as NO_x (a rapidly interconverting combination of NO, NO_2, and NO_3) that generates tropospheric ozone, but in this study the only climate-change effects we track are the direct emissions of greenhouse gases.

Local and regional air pollution: Household combustion of wood or fossil fuels also contributes to local and regional air pollution by emitting NO_x and wood smoke. NO_x is generated partly from oxidation of any N in the fuel itself (esp. in coal) but largely from oxidizing atmospheric N_2. NO_x contributes to acid rain, and can also cause or worsen respiratory and heart conditions both directly and through reacting with other compounds to form ground-level ozone and fine particulate matter ($PM_{2.5}$) (Minnesota Pollution Control Agency 2013b). Residential wood combustion has been identified as a primary source of air pollution in Minnesota, since it produces 36% of Minnesota's $PM_{2.5}$ pollution and 31% of polycyclic aromatic hydrocarbons (PAHs) (Minnesota Pollution Control Agency 2013b). PAHs are carcinogenic, and other volatile organic compounds (VOCs) from wood smoke contribute to ground-level ozone formation. While we do not quantify fluxes of $PM_{2.5}$ or

VOCs for other sources in this study, we do so for wood smoke from home heating in order to examine the distribution and spatial patterns of this primary source of air pollution.

Terrestrial N deposition: NO_x and other reactive forms of N from household fossil-fuel combustion (N_2O_5, HNO_3, and other aerosol and organic N forms) deposit onto vegetation and soil, where they become biologically available and contribute to the increased N deposition in areas such as the Upper Midwest and Northeastern USA (Munger et al. 1996). Terrestrial ecosystems in Minnesota, like in much of the temperate zone, are generally N-limited and thus sensitive to anthropogenic N additions. While low levels of N deposition can act as N fertilizer to stimulate productivity, higher levels can cause soil acidification, decreased productivity, species loss and invasion, and N leaching to ground or surface water (Lovett and Goodale 2011). While P deposition is also increased by anthropogenic activities, especially agriculture, none of the direct output fluxes within our conceptual household boundary contribute to P deposition. As such, we do not track P deposition in this study.

Water pollution: Nutrients from household activities can pollute ground and surface waters via direct deposition onto waterbodies, runoff and leaching from land, and wastewater treatment plant (WWTP) discharge. High concentrations of some N forms are directly toxic: elevated nitrate in drinking water can cause methemoglobinemia, also known as "blue baby syndrome" (US Environmental Protection Agency 2007), and high ammonium concentrations are toxic to fish and other aquatic organisms (Randall2002). The decomposition of organic C compounds consumes oxygen, and organic-C pollution can deplete dissolved oxygen levels severely enough to cause fish kills (Rabalais et al. 2010). Excess nutrients often drive algal blooms that impair water quality by causing bad odor and taste, lower water clarity and dissolved oxygen, and the loss of desirable species (Smith 1998). Over 90% of lakes assessed in our study area of Ramsey and Anoka Counties, MN are categorized as eutrophic or hypereutrophic (Minnesota Pollution Control Agency 2013a). Blooms dominated by cyanobacteria can also produce several compounds acutely and chronically toxic to humans and other mammals. In Minnesota, animal deaths due to cyanobacterial toxins have been documented since the late 1800s and still occur (Lindon and Heiskary 2009), and in 2002 a human death was confirmed in Wisconsin (Weirich and Miller 2014). Nutrient pollution is a concern not only for local lakes and groundwater in the Twin

Cities metropolitan area, but also for downstream waterways on the Mississippi River (e.g. Lake Pepin, MN) and the hypoxic zone in the Gulf of Mexico (Rabalais et al. 2002). Although P is generally considered limiting for freshwater, and N for coastal marine waters, recent studies recommend that controlling both N and P is more effective in reducing algal blooms than controlling only a single element (Elser et al. 2007, EPA Science Advisory Board 2007, Harpole et al. 2011, Howarth et al. 2011) so we consider both N and P as potential pollutants for all waterbodies.

Too Little: Nutrient scarcity

At the same time that excess nutrients are polluting many urban ecosystems, society also faces shortages of essential nutrients. Most household C and P comes from nonrenewable sources: respectively, fossil fuels and phosphate rock. Running out of nonrenewable nutrient sources is not just a distant problem for future generations; depleting the most easily extracted deposits can result in price shocks, international tensions, and other problems (Cordell et al. 2011). Fossil fuels can be replaced with other energy sources, but there is no substitute for P in food production. Because of the uncertainty over the size of remaining P deposits, and the fact that most P deposits occur in only a few countries globally, reducing the reliance on mined P is necessary to ensure sustainable food production (Cordell and White 2013). While most household N is derived from atmospheric N_2, an immense and renewable reservoir, converting it to fertilizer requires large inputs of energy, currently from nonrenewable fossil fuels (Ramírez and Worrell 2006). As such, reducing N fertilizer production also helps alleviate fossil-fuel scarcity.

Just Right? Synergies and tradeoffs

In working towards urban nutrient sustainability, getting things "just right" requires that we move beyond focusing on single elements in isolation, and begin to consider how changes in nutrient consumption and waste management will affect not only all elements but also other important services such as energy and water (Cordell et al. 2011). Only a handful of existing urban budgets simultaneously consider N and P, and most focus primarily on quantifying fluxes and trends rather than assessing future options (Boyden1981, Faerge2001, Leitzinger2001, Morée2013, Liu2014) (Boyden et al. 1981, Faerge et al. 2001, Leitzinger

2001, Morée et al. 2013, Liu et al. 2014). In contrast, two studies have considered synergies and tradeoffs for potential changes in nutrient management. Meinzinger and coauthors (2009) analyzed N and P fluxes in Arba Minch (Ethiopia) to evaluate co-composting and urine-diversion scenarios for upgrading the town's sanitation system. Walker and coauthors (2012) conducted a multi-sectoral systems analysis of the Upper Chattahoochee Watershed (Atlanta, Georgia, USA metro region) and presented a framework that can evaluate synergies and tradeoffs among not only C, N, and P but also water and energy.

In this study, we considered interactions among household C, N, and P fluxes, which are often tightly coupled. Where possible we identified synergistic solutions that could increase sustainability for more than one element. Where this was not possible, we quantified the tradeoffs involved.

Reducing, redirecting, and recycling nutrient fluxes

To reduce household nutrient pollution and the use of nonrenewable nutrient sources, we identified opportunities for three complementary actions: reducing, redirecting, and recycling household nutrient fluxes. We identified specific fluxes that can be reduced, since reducing C, N, and P consumption can directly reduce both pollution and the use of nonrenewables, depending on the source and fate of the particular flux. However, C, N, and P are essential elements, especially for human nutrition; reducing household nutrient consumption must also be complemented with improvements in managing household nutrient outputs. We also quantified opportunities to redirect pollutant fluxes to non-polluting fates, and to reduce the use of nonrenewable nutrient sources by increasing C, N, or P recycling. Recycling P directly reduces the need for mined phosphate rock, and recycling N and C reduces the energy demand (currently from fossil fuels) to manufacture fertilizer, paper, and plastic, and to grow grains for livestock feed.

Previous TCHEP studies focused on reducing the largest household nutrient fluxes in order to reduce nutrient pollution, but not all nutrient fluxes leaving the household cause pollution; some fluxes end up harmlessly in inert forms, and some are recovered and recycled. In this study, we quantified the eventual fates of household nutrient fluxes by tracing C, N, and P outputs through various transformations in the waste streams. This

allowed us to identify specific fluxes that may be a small proportion of total fluxes but cause significant environmental or health impacts, and to guide focused efforts to reduce these specific fluxes. Tracing nutrient fluxes from the household to their eventual fates also allowed us to identify and quantify opportunities where improved nutrient management could redirect pollutant fluxes to non-polluting forms or increase recycling of C, N, and P.

Methods

The Twin Cities Household Ecosystem Project

TCHEP quantified C, N, and P fluxes from single-family households in Ramsey and Anoka Counties, Minnesota, USA, spanning a 55 km urban-to-exurban gradient from downtown Saint Paul (the southern end of Ramsey County) to the northern end of Anoka County. We did not attempt a complete life-cycle assessment of all C, N, and P fluxes from manufacturing to disposal, but focused strictly on direct consumption fluxes associated with household behaviors. We defined household fluxes using a conceptual boundary (*sensu* Baker et al. 2007) that includes nutrient fluxes that take place within the physical parcel boundaries as well as off-property fluxes from household members' food consumption, travel, and home electricity use.

Sampling methods are fully described in Fissore and coauthors (2011) and summarized here: Our sample population was restricted to owner-occupied, single-family, detached homes without wetlands on the property. In May 2008, we mailed a 40-question survey (Nelson et al. 2008) to a stratified-random (by census tract population density) sample of 15,000 homes, and received 3,100 responses (21% response rate). In addition to survey responses on household characteristics, behaviors, attitudes, and norms, we also requested permission to obtain household energy billing records from electricity and natural gas providers (spanning May 2007-April 2008) and to visit the property to measure vegetation. After data entry and error checking, 2763 surveys yielded sufficiently complete information for this study; we also obtained energy records for 1843 of these households. From households where we obtained a completed survey, energy records, and vegetation-measurement permissions, we selected 360 households as a stratified-random subsample for

vegetation measurements. This study uses the largest dataset possible for calculating each individual flux, and we indicate for each result whether it used the 2763, 1843, or 360-household dataset.

The Household Flux Calculator

TCHEP developed a computational tool, the Household Flux Calculator (HFC), to combine survey data (on human diet, pet ownership, driving habits, air travel, recycling, and yard management) with energy records, vegetation management, and county parcel data in order to estimate annual fluxes of C, N, and P in kg of element per household (hh^{-1}) per year. HFC calculations are described fully in Fissore and coauthors (2011, see Appendix A of that paper) and have been expanded to include additional outputs and transformations for this study. In addition, we also corrected and updated the following existing HFC calculations:

Food Waste: Calculations of food-waste fluxes to wastewater (via garbage disposals) used an incorrect value for N of 0.96 kg N $capita^{-1}$ yr^{-1} (Fissore et al. 2011 Table A.10). The correct value from Siegrist (1976) is 0.23 kg N $capita^{-1}$ yr^{-1}, which also changed calculations of total food waste (Fissore et al. 2011 Tables A.10 and A.11). We also fixed a coding error in the HFC which failed to add food waste N fluxes to wastewater outputs for households that used a garbage disposal. At the same time, we updated the food waste stoichiometry calculations with recently published N and P measurements for residential food waste. Food waste element content is now calculated on a dry-weight basis as 44% C (assuming 50% C in ash-free dry weight), 3.3% N, and 0.7% P (Banks et al. 2011). We used the Banks et al. (2011) value of 72.3% moisture content for calculating the amount of food waste entering the landfill, as Beck (2000) reports wet weight of food waste. Annual per-capita food waste fluxes are listed in Table 1.1.

Lawn mower C: We corrected the C flux from gasoline lawn mower use, assuming 20 hr annual lawn mower use (Baker et al. 2007), which consumes 15.8 L of gasoline (Christensen et al. 2001). We then calculated the resultant CO_2, CH_4, NO_x, and N_2O fluxes as described in the original HFC and below.

Electricity NO_x: The original HFC used an incorrect value for NO_x produced during electricity generation (Fissore et al. 2011 Table A.3). We corrected it using EIA data on the total CO_2 and NO_x produced by Minnesota's electric industry in 2008 (US Energy Information Authority 2013) to yield the NO_x-N:CO_2-C ratio in Table 1.2.

Leaf Management: Many survey respondents wrote in their own responses when asked how they dispose of their leaves, which then had to be interpreted as off-site or on-site disposal for the landscape model. The original HFC interpreted responses like "compost site" as an on-property compost pile. We have re-interpreted these to mean an off-property county compost site.

Soil storage: We fixed a coding error in the HFC that did not account for grass-clipping export when calculating changes in soil N and P (soil C calculations were correct), which also affected calculations of inferred N and P fluxes. We now calculate changes in soil N and P using ratios with the change in soil C to match our original assumption that the turfgrass soil C:N:P ratio remains constant (Fissore et al. 2011).

Outputs, transformations, and fates

For this study, we expanded the HFC to track C, N, and P fluxes from the household to their eventual environmental fates. Some output fluxes (garbage, wastewater, compost) undergo additional transformations between leaving the household and being released into the environment. We use "outputs" to describe the fluxes as they leave the household, and "fates" to describe the fluxes as they leave managed waste streams and are released into the environment. For example, C in household garbage (*output*) may be burned in an incinerator (*transformation*) and released to the atmosphere as CO_2 (*fate*).

Airborne Outputs

Fossil-fuel carbon dioxide: We separated fossil-fuel derived carbon dioxide (CO_2) from other sources of CO_2. In the HFC, fossil fuel combustion is tracked for air travel, vehicle travel, lawn mower use, and home energy use (electricity generation, natural gas, propane, and oil, but excluding wood). Home-energy and air-travel fossil fuel CO_2 are calculated using EIA emission factors. For vehicle travel and lawn mower use, the HFC calculates the volume of

gasoline consumed and C content per unit volume. We assumed all C in gasoline is combusted to CO_2, other than the methane calculated below, and did not separate out the fractions emitted as soot, carbon monoxide, or other forms.

Non-fossil-fuel carbon dioxide: Carbon dioxide is also emitted from non-fossil fuel sources in the household: wood burned for home energy, human and pet respiration, soil heterotrophic respiration (which incorporates grass clippings decomposing on the lawn), and on-property leaf decomposition. We assumed all C in wood is combusted to CO_2 except for the methane and wood smoke pollutant fractions calculated below for wood heating. For pet and human respiration, we assumed that all non-fiber C in food is respired as CO_2. The HFC's landscape model calculates soil heterotrophic respiration after Milesi and coauthors (2005), accounting for the household's practices regarding grass clipping removal, N fertilization, and watering. Leaf disposal practices are calculated separately from soil respiration; the HFC assumes leaves disposed of on-property are in steady state, where annual decomposition and CO_2 outputs match annual inputs. Some of the C in garbage, wastewater, and compost is also transformed to non-fossil-fuel CO_2 after leaving the household, as described below.

Nitrogen oxides: High-temperature combustion also oxidizes atmospheric nitrogen into NO_x and N_2O. Because N_2O is a potent greenhouse gas, we tracked it separately from NO_x. The original HFC calculated NO_x-N emissions from electricity generation, in-home natural gas combustion, and air and vehicle travel. For this study, we expanded NO_x-N calculations to include lawn mower use as well as propane, heating oil, and wood combustion. We also added N_2O-N calculations for all these combustion sources. Emission factors are listed in Table 1.2 as ratios with CO_2-C, except motor-vehicle emissions. Motor-vehicle emission factors are published per mile driven, which yields different ratios with CO_2-C for vehicles with different fuel efficiencies. Emissions also vary with model years; NO_x-N emission factors are calculated in the original HFC (Fissore et al. 2011), and we added N_2O-N using EIA emissions factors (US Energy Information Authority 2011).

We did not attempt to quantify landscape sources of N_2O-N, especially because measurements in the vicinity of our study area at Cedar Creek Ecosystem Science Reserve (Clark et al. 2009) found these fluxes to be nearly undetectable. NO_x and N_2O are also

produced during incineration of garbage and sewage sludge, as well as incomplete denitrification during wastewater treatment, but data are not available to relate these to household fluxes. As such, we assigned this N entirely to atmospheric N_2 since we could not estimate the proportion going to NO_x or N_2O.

To analyze spatial patterns in NO_x-N fluxes emitted from the house itself (in contrast with remote fluxes such as air travel, vehicle travel, and electricity generation), we calculated a "home heating NO_x" flux combining at-home combustion of natural gas, propane, oil, and wood. Because the HFC assumes a uniform value for lawn mower use, we excluded the minor contribution of lawn mower NO_x-N from this flux even though lawn mower NO_x-N is also emitted on the property. We refer to this flux as "home heating NO_x" although it includes some non-heating uses such as natural-gas clothes dryers, water heaters, and kitchen stoves and ovens. The human health impact of NO_x is determined not only by NO_x emissions, however, but also by the number of people exposed to it and other factors. We calculated an index of potential human exposure to home-heating NO_x, multiplying each household's home-heating NO_x-N flux by the local housing density (households km^{-2}) in their 2000 US Census "partial block group," a unit of relatively homogeneous population density created by dividing census block groups along municipal and other political boundaries (Hammer et al. 2004).

Methane: We expanded the HFC to calculate methane (CH_4) emissions from all combustion sources; the original HFC only calculated CH_4 for electricity production. We did not attempt to quantify landscape, human, or pet emissions of CH_4. Emission factors are listed in Table 1.2, except motor-vehicle emissions, which we calculated per mile using EIA emissions factors (2011) for different model years, similar to N_2O-N. Because the original HFC tracked all C in fuel inputs for wood and gasoline, we subtracted CH_4-C from the total C fluxes for these fuels before assigning the remainder to CO_2-C. For air travel and non-wood household energy, however, the original HFC calculated CO_2-C using EIA emissions factors, so CH_4-C represents an additional C flux.

Residential wood smoke: Particulate and volatile wood smoke pollution emissions vary substantially depending on wood species, moisture, outdoor temperature, and the type of

wood appliance used (Minnesota Pollution Control Agency 2013b). Since we do not have information on any of those factors, in this study we used uniform emission factors from mixed hardwoods burned in a fireplace: 4.0 g particulate C and 7.1 g non-methane volatile organic C per kg wood , hereafter referred to jointly as "wood smoke C." We also calculated an index of human exposure to wood smoke emitted by our study households, multiplying each wood-heating household's wood smoke C flux by housing density (households km^{-2}) in their partial block group (U.S. Census 2000,Hammer et al. 2004).

Wastewater

Wastewater combines human excreta, food waste (if the household uses a garbage disposal), and toilet paper, detergents, etc. We assume that all N and P in consumed food, as well as all fiber C, are excreted into wastewater (i.e. no gain or loss of human biomass). Wastewater calculations are described fully in the original HFC, except the food-waste corrections detailed above.

Solid Waste

Garbage: The HFC tracks C, N, and P to household garbage from residential and industrial/commercial/institutional (ICI) food waste, non-recycled paper and plastic, and pet waste (scooped dog feces and all cat waste). The HFC does not attempt to account for all C, N, and P in household garbage. For example: wood, carpet, rubber, and textiles combined account for 11% (by mass) of residential garbage entering Twin Cities landfills (Beck 2000), a mass comparable to that of plastic or food waste. While data exist to estimate the elemental composition of these materials, we chose not to include them in the HFC because we do not have data on any household actions or choices influencing them. The bulk of wood and carpet, in particular, are likely to be from a small proportion of houses undergoing construction, remodeling, or demolition. Splitting these equally among the metropolitan population would not accurately reflect most households' outputs.

Recycling: To estimate plastic and paper fluxes to recycling vs. garbage, the HFC allocates newspapers and magazines from each household using the proportion survey respondents

said they recycle. All other paper and all plastic are allocated as fixed per-capita fluxes (Fissore et al. 2011).

Compost: Leaves and grass clippings are assigned to either on-site or off-site disposal in the HFC, based on survey responses about the household's disposal practices and the proportion of their property that they rake. If no information was provided about disposal practices, we assumed that leaves are disposed of on-site. We assigned all off-site disposal to county composting facilities, as it is illegal to dispose of yard waste in household garbage. On-site disposal is already accounted for within the HFC landscape model (Fissore et al. 2011, 2012) rather than a separate on-site yard waste compost flux.

Food compost, by contrast, is separate from the HFC landscape model. If a household reported that they compost food waste, whether or not they have a garbage disposal, we assigned all residential food waste to backyard food compost (none of our study households were served by municipal composting programs).

Food waste to livestock: We expanded the HFC to include ICI food waste that is recovered for direct feeding to livestock or as an ingredient for manufacturing processed livestock feed (hereafter referred to jointly as food-to-livestock). As with landfilled food waste, we calculated annual per-capita fluxes (Table 1.1) by dividing the estimated total weight of food-to-livestock (Solid Waste Management Coordinating Board 2007) by the population of the metro-area solid waste district, and multiplied by the C, N, and P concentrations in food waste described above (Banks et al. 2011). This represents an additional food-waste flux not already in the HFC, as the original calculations are based on measurements of food waste entering the landfill, and increases per-capita food waste estimates by 44%.

Landscape Losses and Storage

Based on the mass-balance of N and P inputs and outputs, the HFC landscape model calculates an "inferred flux." Positive values (where inputs exceed outputs plus storage) represent excess nutrients likely to be lost from the landscape. We refer to positive inferred fluxes as "landscape losses" in this study. Although the model cannot partition this flux among loss pathways (e.g. denitrification, erosion, runoff, leaching to groundwater), we

regard landscape losses as potential sources of pollution. Negative inferred flux values represent unmet ecosystem demand, which may cause "mining" of soil nutrients, reduce growth or accumulation rates, or favor N-fixing species (e.g. clover) in the lawn.

Landscape storage is calculated in both soil organic matter and tree wood. Negative soil accumulation values can result when lawn management practices result in net decomposition of soil organic matter, releasing the C as non-fossil-fuel CO_2 and mineralizing the N and P for plant uptake or loss. Unlike in earlier analyses (Fissore et al. 2011, 2012), we set negative inferred flux and soil accumulation values to zero before summing component fluxes or averaging across households. Negative inferred fluxes are not meaningful for this study's goal of estimating potential nutrient pollution; they do not offset other landscape output fluxes nor losses from other parcels. For soil accumulation, the overall net change in residential-landscape soil nutrient pools has already been calculated in earlier studies by including negative accumulation values (Fissore et al. 2011, 2012). For this study, we instead chose to set negative accumulation values to zero to calculate gross accumulation fluxes and track the fate of nutrients that entered the landscape from other sources (e.g. fertilizer, atmospheric deposition, pet waste, weathering) and compare the amounts that are accumulated, lost, or composted.

Transformations and Fates

Wastewater: The transformations and eventual fates of wastewater are very different for households connected to sanitary sewers or to individual septic systems. To determine whether a household was on sewer or septic, we combined survey responses with sewershed maps and municipal records. For unclear cases, we assigned the household to sewer, as 90% of the Twin Cities metropolitan area population is served by sewer (Metropolitan Council 2014).

Sewer-connected households in our study were primarily served by the Metropolitan Wastewater Treatment Plant (WWTP) in Saint Paul, MN. A few study households were served by a small community sewer for which nutrient-removal data are not available; we calculated these households' transformations as if they were served by the Metropolitan WWTP. Using the Metropolitan WWTP's average influent and effluent concentrations for

2008 (Metropolitan Council, *unpublished data*), we calculated removal efficiencies of 94% for C (measured as chemical oxygen demand), 98% for N (only ammonium was measured; we assumed all forms of N are removed similarly), and 92% for P (measured as total P). In addition to decomposition and denitrification in the treatment tanks, elements are removed in scum and sludge that are dewatered and incinerated on site. We assumed that all removed C is decomposed or combusted to non-fossil-fuel CO_2, and all removed N is denitrified or combusted to N_2. Data are not available to estimate the proportion converted to CH_4, N_2O, or NO_x. All removed P is retained in the incinerator ash and landfilled. The remaining C, N, and P in the WWTP's effluent are released to the Mississippi River.

For septic-connected households, we assumed that all wastewater enters their septic system, though for many household members a portion of their excretion takes place away from home and therefore enters other wastewater systems (other septic systems or the sewer system). Some wastewater C, N, and P is periodically pumped out of the septic tank as septage, and the remainder enters the drainfield in septic effluent. We assume that decomposition and denitrification in the septic system itself are negligible; although septic tanks are often conceptualized as anaerobic digesters, studies have not found significant differences between influent and effluent total C concentrations (Diak et al. 2012).

We calculated septage fluxes of 22.7 kg C, 1.1 kg N, and 0.40 kg P hh^{-1} yr^{-1}, based on 1893 L/yr (nominally 500 gal/yr, assuming an average 1,500 gal tank pumped every three years per Minnesota Pollution Control Agency guidelines) and septage concentrations from EPA (1994). Some study households had wastewater element fluxes less than these average septage fluxes, in which case we assigned the entire wastewater flux of that element to septage. Septage fluxes are assigned to land application (primarily forage crops), although some septic pumpers dispose of septage to WWTPs.

Household garbage: Household garbage in our study area is either landfilled directly or processed for waste-to-energy incineration. We do not have enough information to assign individual households to either landfill or incineration, as many municipalities in our study area are served by multiple garbage haulers, so we split each household's garbage between landfill and incineration according to county averages. Ramsey and Anoka Counties both

landfilled about 40% of their unrecycled mixed municipal solid waste in 2010, and processed 60% for incineration (Anoka County 2012, Ramsey County 2012). For incinerated garbage, we assumed that all C is combusted to CO_2 and N to N_2; data are not available to estimate the proportion converted to CH_4, N_2O, or NO_x. All incinerated P is retained in the ash and landfilled. Although some of the C in household garbage ultimately derives from fossil-fuel feedstocks (e.g. plastics), we assign all incinerated C to non-fossil-fuel CO_2 in order to reserve the "fossil-fuel CO_2" category for CO_2 from fossil fuels burned directly by household activities: air and car travel, household energy (whether off-site electricity generation or at-home combustion), and lawn mower use.

Compost: We assumed that all compost enters a managed compost pile, whether at home (food compost) or at county compost facilities (leaves and grass clippings). We assumed that these compost piles are managed to minimize nutrient losses; all N and P are therefore retained in the finished compost. 50% of the C, by contrast, is lost during composting as non-fossil-fuel CO_2 (University of Minnesota Extension 2008). We assigned finished compost to land application, primarily in residential or community gardens.

Identifying opportunities for improvement

We used our results to identify two broad classes of fluxes for detailed examination in this study. The first is pollutant fluxes for focused consumption-reduction efforts. Previous TCHEP studies have identified and analyzed household activities generating the largest nutrient input fluxes, and we built on their findings and recommendations by using our results to examine smaller important pollution fluxes that have not yet been addressed. For example, we did not reiterate previous findings on the distributions and drivers of the largest household C fluxes, fossil-fuel CO_2-C from air and car travel and household energy (Qian et al. *in review*, Fissore et al. 2011), but we evaluated whether including CH_4 and N_2O in household greenhouse-gas emissions leads to different findings and recommendations (see *Discussion: Reducing specific pollutant fluxes - Global climate change*). This study also examined spatial patterns and differences in waste management among households, which we used to identify new subsets of households where focused efforts are likely to yield the greatest reductions in nutrient pollution.

Second, we went beyond demand reduction and used our results to identify opportunities where improved waste-management practices could redirect pollution fluxes to non-polluting forms or increase recycling. We modeled scenarios of these improved practices, assuming unchanged household nutrient consumption but with different practices either for disposing of household wastes or managing wastes after they leave the household. In order to quantify the potential benefits of improving waste-management practices, we calculated the changes if all households were to participate. Even though 100% participation is not likely to be realistic for all scenarios, these calculations provide useful comparisons and a basis for refined predictions that can incorporate forecasts of household participation.

Statistical Analyses

Because household nutrient fluxes follow skewed distributions (Fissore et al. 2011), for output fluxes we calculated summary statistics that do not assume a normal distribution: mean, median, first and third quartiles (Q1 and Q3), and two measures of dispersion: the median absolute deviation (MAD) and the Gini coefficient. MAD is the median value of all data points' absolute deviations from the dataset median, multiplied by 1.48 to adjust it such that, for a normal distribution, MAD equals the standard deviation. Like the standard deviation, MAD is a measure of dispersion with the same units as the data themselves, but it does not assume a normal distribution and is robust against outliers. The Gini coefficient (G) is a unitless measure, most often used to express income or wealth inequality (Ceriani and Verme 2012), but here used to quantify the disproportionality among study households. A Gini value of zero would reflect perfect equality (all households contribute equal nutrient fluxes), increasing to a value of one reflecting the case where a single household contributes 100% of the nutrient flux. The Gini coefficient is calculated based on the Lorenz curve, a variant of which was used to graph disproportionality among our study households' input fluxes (Fissore et al. 2011).

We did not compute similar summary statistics when expressing nutrient fluxes by fate, since some fates combine component fluxes with different sample sizes. For example, non-fossil-fuel CO_2 was a combination of fluxes from the household landscape (n = 360), human

and pet respiration (n = 2763), and other sources. We calculated the total mean flux to each fate as the sum of the means of individual component fluxes.

To examine spatial patterns in home-heating NO_x, wood smoke, and their exposure indices, we computed simple linear regressions with housing density (of the partial block group) or distance from the nearest central business district (i.e. downtown Minneapolis or Saint Paul). All statistical analyses were performed in R 3.0.2 (R Core Development Team 2013), using base packages plus the `gini()` function in the reldist 1.6-2 package (Handcock 2013).

Results

Overview

To provide context for our results on specific fluxes and waste-management scenarios, we first present an overview of C, N, and P fluxes and discuss the overall range of disproportionality for different household output fluxes. We group output fluxes by household components to match the original TCHEP study (Fissore et al. 2011), and then track them from the household to our three fate categories: pollution, inert forms, and recycling. Note that total nutrient fluxes reported here differ somewhat from those originally reported by Fissore and coauthors (2011) due to expansions and corrections to the HFC, as well as the use of larger datasets (only the 360 households with vegetation measurements were used in the original study).

Carbon: Household output fluxes of C were 8126.7 kg hh^{-1} yr^{-1}, equal to the amount of C in 12706 L of gasoline or 12.8 cords (46.6 m^3) of firewood. Most C output fluxes (Table 1.3a, Figure 1.1) were from air and car travel (43%) and household energy (41%). Among the remaining C fluxes, the component with the largest C output fluxes was the household landscape (9.5%), followed by human diet (3.1%) and paper & plastic (2.6%). The fate of most household C was pollution (Table 1.4): fossil-fuel CO_2 accounts for 83% of all household C fluxes, primarily from air and car travel and household energy, plus a minor contribution (0.01%) from lawn mower use. Sixteen percent of household C ended up in inert forms, primarily non-fossil-fuel CO_2 (11%) and landscape storage (3.5% in wood, 0.5%

in soil). Only 1.4% of C was recycled as paper and plastic (0.8%), land application (0.4%), or food-to-livestock (0.1%).

Nitrogen: Household output fluxes of N were 54.6 kg hh^{-1} yr^{-1}, equal to the amount of N in 237 kg of lawn fertilizer (assuming 23% N by weight, e.g. a 23-0-6 fertilizer), most readily envisioned as 13 bags each weighing 18 kg (nominally 40 lbs). Major N output fluxes (Table 1.3b, Figure 1.2) were fairly evenly split among human diet (29%), household landscape (28%), and air and car travel (26%). While C fluxes were similar between travel and energy, N fluxes for air and car travel were almost three times those for household energy, reflecting "dirtier" combustion in engines than heat- or electricity-generation (Table 1.2). The majority of N fluxes also caused pollution (54%, Table 1.4): 19% were in forms that can lead directly to water pollution, and 35% were airborne pollutants that contribute to global climate change or local/regional air pollution and N deposition. Inert forms of N were also substantial (40%), primarily as N_2 (30%) and landscape storage (2.9% in wood, 5.9% in soil). Only 5.2% of N was recycled as land application (3.6%) or food-to-livestock (1.6%).

Phosphorus: Houshold output fluxes of P were 4.7 kg hh^{-1} yr^{-1}, equal to the amount of P in 47 kg of garden fertilizer (assuming 10% P by weight, e.g. a 5-10-5 fertilizer), most readily envisioned as 2.6 bags each weighing 18 kg (nominally 40 lbs). The majority of P output fluxes (Table 1.3c, Figure 1.3) were split between "detergents etc." (32%) and human diet (31%), followed by household landscape (20%) and food waste (12%). Most P ended up as inert forms (78%, Table 1.4), primarily because household garbage and most household wastewater P were eventually disposed of in the landfill (64%) either directly or as incinerator ash. Modern landfills are designed to seal materials in and prevent seepage, so we considered landfill disposal to be an inert fate. Thirteen percent of P fluxes were in forms that can cause water pollution, either immediately near the household (6.1% from septic effluent, 1.9% from landscape losses) or in the Mississippi River as WWTP effluent (4.6%). P recycling was higher than for C or N, with 9.2% recycled via land application (5.4%) or food-to-livestock (3.8%).

The largest P output flux, "detergents etc.," also has significant uncertainty. We estimated "detergents etc." fluxes using data from a single study of four rural households in 1976

(Siegrist et al. 1976). To our knowledge, this is the only empirical study to measure C, N, and P fluxes from different components of a household's wastewater stream. We used their greywater (non-toilet) data to estimate household fluxes from detergents and other products that leave the home in wastewater (Fissore et al. 2011). We subtracted laundry P to reflect the subsequent ban on laundry-detergent P, but there are likely to have been additional, unknown changes in the composition or use of various products between 1976 and 2008; further restrictions on P in automatic dishwasher detergents began in 2010, after our survey. Furthermore, it is not known how similar these four households in rural Wisconsin might be to our study households. Because we estimated that detergents and other products are a major P source that may account for almost a third of household P outputs, we highlight the need for new studies to characterize greywater nutrient fluxes from modern households.

Disproportionality among households: Total output fluxes for each HFC component are similar to input fluxes, the distributions of which are described in detail by Fissore and coauthors (2011 and 2012). All individual output fluxes have right-skewed distributions to varying degrees (Table 1.3). To quantify the disproportionality among households, we calculated the Gini coefficient for each flux, where 0 is perfect equality and 1 is complete inequality (see *Methods: Statistical Analyses*). For reference, fluxes calculated as fixed per-capita values (e.g. "detergents etc." or food-to-livestock) had G = 0.26; this disproportionality was due solely to variation in the number of household members. Household energy CO_2-C outputs are noteworthy for having a lower G = 0.19. The highest disproportionality among households was found in fluxes where only a small proportion of households participated in a given activity, such as wood-heating (4.7% of households, G = 0.97) or backyard food composting (18% of households, G = 0.86). Landscape fluxes also exhibited high disproportionality, driven largely by differences in property size; landscape fluxes are discussed in depth in Fissore and coauthors (2012).

Interpreting "per-household" values: Although we reported flux values per household per year, the overall averages should not be interpreted as representing a realistic or "typical" single household's fluxes because no single household's fluxes had the depicted sewer/septic or landfill/incineration splits. Fluxes generated by only some households (e.g. wood smoke C) were averaged over the total number of study households, not the smaller number of

households generating that flux (unless otherwise specified for a given analysis). The average flux values reported here (as in all TCHEP studies) are most accurately interpreted as total fluxes for the population of single-family owner-occupied households in Ramsey and Anoka Counties, scaled to per-household units for easier comprehension.

Current pollution fluxes

We quantified household C, N, and P fluxes that cause pollution by contributing to global climate change, local and regional air pollution, terrestrial N deposition, or water pollution. Some of the largest pollutant fluxes, fossil-fuel CO_2-C, NO_x from air and car travel, and landscape N losses, have already been addressed in detail in previous TCHEP studies. In this study, we identified additional pollutant fluxes that may not be a large fraction of total C, N, or P fluxes but still have substantial impacts on ecosystem functions or human health. Tracking nutrient fluxes to their eventual fates also allowed us to directly examine the disproportionality among households' pollution fluxes, which in some cases is much larger than the disproportionality among their input fluxes, and thereby identify additional opportunities where focusing consumption-reduction efforts on a subset of households could yield substantial reductions in nutrient pollution.

Global climate change: For the 1843 households with energy records, we calculated their contributions to 100 yr global warming potential (GWP_{100}) from CO_2, CH_4, and N_2O emissions from household energy, car travel, air travel, and lawn mower use (Table 1.5). (Note that these fluxes for air and car travel differ slightly from those in Table 1.3, where we used the full 2763 dataset for those fluxes.) Almost all household GWP was from CO_2 (99%), with minor contributions from CH_4 (0.2%) and N_2O (1.0%). Comparing among activities, GWP was mainly produced from home energy (47%) and vehicle travel (43%), plus 9.3% from air travel and 0.1% from lawn mower use.

For several households, wood heating was a substantial fraction of total household GWP, in some cases >10%. The CO_2 released from wood combustion is considered to be GWP-neutral, as that carbon was taken up from the atmosphere during the trees' lifetime. However, residential wood combustion produces much more CH_4 and N_2O per unit heat than other home energy sources (Table 1.2). We estimated that residential combustion of

one cord (3.6 m^3) of wood has a GWP equivalent to 51.3 kg of CO_2-C, which is the amount of C in 80.3 L of gasoline or 9.4 Ccf of natural gas. The Minnesota Energy Assistance Program estimated that heating a three-bedroom house (roughly 110-140 m^2) entirely with wood requires 7.5 cords of wood annually (Minnesota Energy Assistance Program 2013), which would produce 25% of the GWP produced by our study households' average natural-gas usage, 28% of the GWP produced by average propane usage, or 19% of the GWP produced by average heating-oil usage (averaging together only houses that use each heating source).

Local and regional air pollution: Unlike greenhouse gases, which become well-mixed in the atmosphere and have global effects, the health and environmental impacts of air pollutants such as NO_x, particulate matter, and volatile organic compounds are much more localized. Here, we separated out NO_x-N fluxes that are emitted in different places, and described the spatial patterns of emissions and exposure for home-heating NO_x-N and wood smoke C.

1. NO_x: Household activities emitted NO_x-N fluxes at several different spatial scales: air travel fluxes (2.8 kg N hh^{-1} yr^{-1}) were a mix of ground-level hotspots at airports and high-altitude emissions spread along regional to global flight paths; car travel fluxes (10.9 kg N hh^{-1} yr^{-1}) were ground-level and mostly regional, concentrated along major roads; electricity generation fluxes (3.5 kg N hh^{-1} yr^{-1}) were localized at various power plants in the region; and home heating (1.4 kg N hh^{-1} yr^{-1}) and lawn mower fluxes (0.02 kg N hh^{-1} yr^{-1}) were both generated at the house itself.

For our study households with energy records (n=1843), the mean home-heating NO_x-N was 1.4 kgN hh^{-1} yr^{-1}, which was 13% of these households' mean vehicle NO_x-N (the largest source of NO_x-N). Home-heating NO_x-N fluxes showed moderate disproportionality (G = 0.22), and as expected were positively correlated with house size. Our exposure index (combining both emissions and population density) showed much higher disproportionality (G = 0.49), driven by greater variability in population density than in home-heating NO_x-N fluxes, and 10.4% of households generated 50% of potential NO_x exposure. Households in densely populated areas had the highest index for potential human NO_x exposure: of the

10.4% of households that generated 50% of potential human exposure, 91% were within the city limits of Saint Paul.

Home-heating NO_x-N fluxes were negatively correlated with distance from the nearest CBD, and positively correlated with population density (both weak but statistically significant), which is the opposite spatial pattern from house size or total home-energy CO_2-C fluxes. This spatial pattern in home-heating NO_x-N fluxes may be driven by greater use of electricity for heating in households further out of the city, since electric heating does not produce NO_x at the house itself. A logistic regression found that the likelihood of using natural gas, oil, and/or propane showed a slight but statistically significant decrease with increasing distance from the nearest CBD ($p=2.2x10^{-9}$).

We also examined households with home-heating NO_x-N greater than twice the mean (57 households, 3.1%) to understand their NO_x-N sources. We expected that high home-heating NO_x-N fluxes would be driven primarily by wood heating, but this only explained some of the high emitters. A majority of high-emitter households (54%) did not burn any wood, and an additional 13% had wood NO_x-N fluxes lower than other home-heating NO_x-N fluxes. While propane and oil produce more NO_x per unit heat than natural gas, they were not significantly over-represented among high emitters. High home heating NO_x-N fluxes can be partly explained by the use of wood heat, and partly by higher heating usage overall.

2. Wood smoke: Of our 2763 study households, 131 (4.7%) reported burning wood for heat. Our data are comparable to a smaller MN-DNR survey of wood burning, which estimated that 4.2% of households in the seven-county Twin Cities metro area burned wood for heat in the 2007-2008 heating season, almost all as a secondary heating source (Barzen et al. 2008). They also found that 85% of wood-burning households in the metro area burned wood for pleasure or camping, usually in small volumes, which we did not ask about in our survey.

Within our wood-heating households, wood smoke C exhibited strong disproportionality (G = 0.43). Over half of all heating wood smoke C was emitted by households burning four or more cords per year (19% of wood-heating households). Using the MN-DNR data to include smoke from wood burned for pleasure, we estimated that 25% of all residential

wood smoke C in the metro area was generated by 2.9% of wood-burning households. Our exposure index showed a stronger disproportionality (G = 0.57), and 14.5% of wood-heating households generated 50% of heating-related wood smoke exposure. (The MN-DNR data lacks the spatial resolution to calculate exposure indices, so we could not include wood burned for pleasure in the exposure calculations.) Households in densely populated areas had the highest index for potential human exposure to wood smoke, even if they only burned a small volume of wood. While the volume of wood burned showed a weak increase with distance from the nearest CBD ($R^2 = 0.10$, $p = 0.0002$), the decrease in population density was much stronger and led to wood smoke exposure decreasing with distance from the CBDs ($R^2 = 0.57$, $p = 2x10^{-16}$). Of the 14.5% of households that generated 50% of potential human exposure, almost half were within the city limits of Saint Paul; one-third burned only one cord of wood per year.

Terrestrial N deposition: NO_x-N emissions not only impact human health, but also contribute to terrestrial N deposition. For comparison, the average NO_x-N emission from our study households (18.6 kg N hh^{-1} yr^{-1}) was equal to the amount of atmospheric N deposition over 1.7 ha, as measured at Cedar Creek Ecosystem Science Reserve at the exurban end of our study area (N deposition data are not available for elsewhere within the Twin Cities metropolitan area). This flux of atmospheric deposition adds the same amount of N as in 81 kg of lawn fertilizer (assuming 23% N by weight), most readily envisioned as 4.5 bags weighing 18 kg (nominally 40 lbs) each. Most of our study households' NO_x-N was from car travel (10.9 kg N hh^{-1} yr-1), which primarily deposits within 100 m of the road (Bettez et al. 2013). Our survey did not include any information on where household members drove, so we were not able to estimate which particular ecosystems were most impacted by our study households' N deposition.

Water pollution: In addition to causing air pollution and terrestrial N deposition, the 18.6 kg hh^{-1} yr^{-1} flux of NO_x-N was also the largest flux of potential water pollution. NO_x-N can deposit directly onto waterbodies, and N deposited on land can leach or run off to ground and surface waters. Landscape losses, septic effluent, and WWTP effluent contributed an additional 1.9 kg C hh^{-1} yr^{-1}, 10.5 kg N hh^{-1} yr^{-1}, and 0.59 kg P hh^{-1} yr^{-1} in dissolved forms that are also potential water pollutants. We classified dissolved organic C in WWTP effluent as a

pollutant, since it contributes to oxygen depletion in the Mississippi River, whereas C in septic effluent is generally not considered to be a pollutant and may be beneficial for supporting denitrification (Greenan et al. 2006). We also classified P as a potential pollutant leaching through soil to groundwater (for landscape losses and septic effluent), even though phosphate adsorbs strongly to soil particles and is frequently considered minimally leachable. Much of our study area is on sandy, P-rich soils, and there is increasing evidence for substantial P leaching to groundwater, especially in the Upper Midwest (Beauchemin et al. 1998, Haygarth et al. 1998, Hooda et al. 1999, Burkart et al. 2004, Djodjic et al. 2004).

Current nutrient recycling

Household nutrient recycling can be divided into two broad categories: paper and plastic recycling, and recycling to the food-production system (i.e. compost, septage application, and food-to-livestock). We estimated that our study households recycled 64.6 kg C hh^{-1} yr^{-1} in paper and plastic, but this is likely an overestimate. Our data showed a large mismatch between self-reported paper recycling rates (our survey did not ask specifically about plastic) and values reported by recycling companies and landfill studies. Averaging together survey responses (n=2763) predicted an overall 91% paper recycling rate. Excluding newspapers and magazines, 50.4 kg $capita^{-1}$ yr^{-1} of other recyclable paper were landfilled from metro-area residences (Beck 2000), while only 8.5 kg $capita^{-1}$ yr^{-1} were recycled (Eureka Recycling *personal communication*, in Fissore et al. 2011). (We excluded newspaper and magazines because their sizes and circulation rates are likely to have changed substantially between the 1999 landfill study and our 2008 survey.) Even if recycling rates were much lower in 1999 than 2008, these data cannot be reconciled with the self-reported 91% recycling rate.

Our study households also recycled 45.9 kg C, 2.8 kg N, and 0.43 kg P hh^{-1} yr^{-1} back into the food-production system, with the majority via land application of septage and compost. For comparison, this was 14% and 17% as much N and P, respectively, as household fluxes of human and pet food (including consumed food and food waste). Our estimates of compost and septage recycling represent upper bounds for several reasons: If a household reported composting food waste, the HFC assigns all residential food waste to compost, although some food waste is likely to be disposed of by other means (deliberately or

accidentally). Furthermore, there may be a gap between self-reported and actual food composting rates, similar to recycling. Once food waste or yard waste is composted, we assumed all compost is used to grow food, although in reality some is applied to other portions of the landscape (e.g. ornamental gardens, mulching). Similarly, not all septage is applied to cropland; some septic pumpers in our study area dispose septage via WWTPs, and some may spread septage on non-crop land. Finally, some of the recycled nutrients applied to cropland or gardens may be wasted; for example, we expect that some residential gardeners essentially "overfertilize" with compost, applying it in excess of what their crops actually need.

Discussion

By tracking household C, N, and P fluxes to their eventual fates, we identified several pollutant fluxes that exhibited strong disproportionality and are likely amenable to focused consumption-reduction efforts. We found opportunities for redirecting only a small fraction of pollutant fluxes, but substantial opportunities to increase nutrient recycling. Our study households have the potential to recycle more N and P to the food-production system (via food-to-livestock and land application of septage and compost) than the amounts of N and P they consume in human and pet food.

Opportunities to reduce nutrient fluxes causing pollution

Global climate change: Previous TCHEP studies have described the distribution and drivers of fossil-fuel CO_2-C fluxes, and identified opportunities for focused source-reduction efforts. Air and car travel showed high disproportionality among households, indicating opportunities to substantially reduce emissions by focusing on high-emitter households (Fissore2011), while CO_2-C fluxes from household energy use were more nearly normally distributed. A subsequent study (Qian et al. *in review*) examined the drivers of electricity and natural-gas C fluxes and tested direct and indirect effects of demographic factors, physical house characteristics (size, age), heating and cooling systems, energy-efficiency actions, and temperature-setting behaviors. Their findings highlighted opportunities to reduce C fluxes by

lowering indoor temperatures in winter, retrofitting older homes, and providing education on practical knowledge and skills to conserve energy.

In this study, we expanded the HFC to include additional greenhouse gases and examined whether previous findings also held for overall household GWP. We found that fossil-fuel CO_2 accounted for 99% of GWP, and the disproportionality among households' GWP was essentially the same as when considering CO_2-C fluxes alone for home energy, vehicle travel, or air travel. We consider the previous studies' CO_2-C findings to be generally robust for overall household GWP, especially because they focused on opportunities to reduce consumption (flying, driving, or home-energy use), which would reduce all greenhouse gas emissions from fossil-fuel combustion. We found noticeable differences, however, for the small fraction of households that burned large volumes of wood, and recommend that any efforts working with these households include the GWP from wood heating as well as fossil-fuel combustion.

Local and regional air pollution: The largest flux causing air pollution was NO_x-N from air and car travel, both of which showed highly skewed distributions very similar to fossil-fuel CO_2-C fluxes from these sources. Because the disproportionality in travel fluxes was addressed in the original TCHEP study, as were opportunities to reduce flying and driving (Fissore2011), in this study we focused on wood smoke C and home-heating NO_x-N, including the spatial patterns of these fluxes and their potential human exposure.

Although wood smoke C accounted for <0.1% of total household C fluxes, residential wood smoke contributes a significant proportion of Minnesota's particulate and volatile air pollution and is an important pollutant flux to focus on (Minnesota Pollution Control Agency 2013b). The strong disproportionality we identified among wood-heating households presents an opportunity to work with a small proportion of households responsible for a large fraction of wood smoke C emissions or potential human exposure. Nearly all of these wood-heating households already used an additional source of heat, which could allow them to burn less wood on days when air quality is impaired. We recommend further studies to characterize these high-emitting households' choices in wood

type and wood-burning appliances, as well as their economics, norms, and values around wood heating, to inform focused efforts to reduce residential wood smoke pollution.

TCHEP provides a unique opportunity to examine spatial patterns in wood smoke C and home-heating NO_x-N fluxes, and to consider how local population density also influences the potential human exposure from our households' emissions. Our focus on home-heating emissions is in contrast with most spatially explicit studies of NO_x pollution, which focus on localized traffic sources and/or regional overviews for ground-level ozone formation. To our knowledge, no other study has looked at this detail of spatial patterns for wood smoke; the MN-DNR study only categorized households by multi-county regions. For both wood smoke C and home-heating NO_x-N, our exposure indices suggest that spatial factors are very important in determining potential human exposure. Differences in local population density were much greater than differences in wood smoke C or NO_x-N fluxes, and as a result the exposure indices showed substantially greater variability and disproportionality among households than did the fluxes themselves. Our data suggest that reducing potential human exposure can be best achieved by focusing not on the highest emitters, but rather on households in densely populated neighborhoods.

It should be emphasized that our exposure indices are rough approximations using population density for the entire partial block group. A detailed calculation of human-health effects would need to include local airshed delineations and small-scale population densities, as well as non-additive and threshold relationships between air pollution concentrations and health effects. Our results do, however, highlight the need for detailed spatial analyses to identify households whose emissions have the greatest impact on human health.

Water pollution: Efforts to reduce detergent and food-waste N and P fluxes to wastewater can yield the greatest reduction in water pollution if they focus on households served by septic systems, which remove a substantially lower proportion of N and P than does the Metropolitan WWTP. We estimated that an average 6% of N and 13% of P were removed from wastewater entering a household septic system, compared with 98% of N and 92% of P entering the Metropolitan WWTP. Only 12% of our study households were served by septic systems, and the focus can be narrowed further by identifying septic-served areas with

high hydrologic connectivity to vulnerable waterbodies (e.g. permeable soils, shallow water table, proximity to surface water). We recommend characterizing the behaviors, economics, values, and norms of this small group of households in order to design effective wastewater-flux reduction efforts.

While wastewater fluxes accounted for 85% of P water pollution, they contributed only 7% of potential N water pollution. Landscape N losses were nearly four times as large, and a number of previous TCHEP studies have examined yard-care practices to help reduce these. The original TCHEP study found high variability among households' fertilization choices, indicating flexibility and opportunities for change (Fissore et al. 2011), and a subsequent behavioral survey found that fertilization choices were driven by a combination of factors, with no single "magic bullet" change point (Martini et al. 2013). Structural aspects and sociodemographics are harder to change, but that study also highlighted the influence of norms, knowledge, and perceived ability – all prime areas for intervention. Knowledge about the details of one's own yard-care practices (e.g. fertilizer N:P:K ratio, mower blade height) was also positively correlated with the likelihood of following additional best management practices that help reduce N pollution (Martini and Nelson *in press*). To help design interventions, Dahmus and Nelson (2014) conducted small-group discussions with residents and found that ecosystems concepts of biotic and abiotic interactions in yards and linkages across yards were particularly salient in describing their yards, while prominent gaps included ecological cycles (e.g., nutrient cycles and food webs), biodiversity, and ecosystem services. Discussion participants were also more likely to share yard-care information with their neighbors than were other residents who only received the same information by mail (Martini et al. *in press*). Our findings in this study reinforce the importance of these efforts to improve residential yard-care practices and reduce landscape N losses, since landscape N losses are a major source of potential water pollution.

NO_x-N fluxes were more than twice as large as landscape N losses, but unlike landscape losses these generally have not been addressed in water-quality efforts. Even if only a fraction of household NO_x-N fluxes were to eventually reach waterbodies, NO_x-N would still be a substantial contributor to water pollution. We recommend that water-quality efforts expand their scope to address this flux in areas where NO_x-N deposits on vulnerable

waterbodies or their watersheds, especially where stormwater or other hydrologic connections readily transport deposited N from land to water. Air-quality efforts do work to reduce NO_x-N emissions, but with a focus on human exposure; we expect that their focus may place a low priority on some NO_x-N emissions that impact few people but cause significant water pollution (for example, a power plant upwind of a sparsely-populated area with vulnerable waterbodies). A better spatial understanding of NO_x-N deposition will help focus efforts to reduce both water pollution and terrestrial N deposition.

Synergies and tradeoffs: Contrary to our expectations, we found few significant synergies and tradeoffs when examining opportunities to reduce pollution by reducing household nutrient consumption. We identified a tradeoff where reducing local air pollution by reducing wood heating increases greenhouse-gas emissions. Reducing volatile and particulate wood smoke C pollution (and the NO_x from wood heating) would require replacing wood heating with fossil-fuel-based heating that produces substantially higher GWP per unit heat. However, focusing efforts on households in densely populated areas is likely to yield the greatest reductions in human exposure to wood smoke C per unit of increased fossil-fuel consumption.

We also identified important differences in the synergistic benefit where reducing fossil-fuel combustion reduces GWP and also NO_x-N fluxes that contribute to local and regional air pollution, terrestrial N deposition, and water pollution. For motor vehicles, this is fairly straightforward: older vehicles generally have greater NO_x-N, CH_4-C, and N_2O-N emissions per mile, because allowable emissions were higher and catalytic-converter and other technologies were not as effective. When reducing fuel consumption, such as reducing driving or upgrading to a more fuel-efficient vehicle, reductions in local air pollution per unit GWP reduced will be higher if efforts are focused on older (or poorly maintained) vehicles. This is consistent with analyses of vehicle scrappage programs (financial incentives to replace older, less efficient vehicles sooner than would be optimal based on private costs alone), which found that replacing vehicles every 5-7 years would minimize NO_x emissions (Spitzley2005).

For household fossil-fuel energy, the synergistic benefits are more complex because different types of household energy produce not only different amounts of NO_x-N per unit GWP, but also in different locations. Because of these differences, we recommend considering this synergy explicitly when deciding how to prioritize efforts to reduce home-energy GWP. Electric generation produces the most NO_x-N per kg CO_2-C, 2.4 times as much as natural gas; propane has the next-highest ratio, 1.5 times as much as natural gas; and oil is comparable to natural gas. However, the location of NO_x-N emissions must also be taken into account. For households in areas of low population density with low potential for local human exposure, we expect that maximizing overall NO_x-N reduction is more important than focusing specifically on home-heating NO_x-N; this can be best achieved by focusing primarily on reducing electricity use, and secondly on home heating in propane-burning households. In contrast, we expect that reducing local emissions of home-heating NO_x-N is more important for households in densely populated neighborhoods, where potential human exposure is highest. Here, efforts to reduce GWP from household energy use should prioritize home heating over electricity.

Opportunities to redirect pollutant fluxes to non-polluting fates

We found few opportunities to redirect pollutant fluxes to non-polluting fates, which highlights the fact that significant reductions in nutrient pollution require changing household behaviors to reduce nutrient consumption. Most pollutants are released from the households in their final form (e.g. fossil-fuel CO_2) and have no potential intervention points downstream of the household. Wastewater fluxes, however, have two opportunities for intervention: reducing food waste entering wastewater, and shifting from septic to sewer.

Opportunity #1: Eliminate garbage-disposal use. Eliminating food waste entering wastewater would redirect this flux from pollution to either inert forms (via garbage) or nutrient recycling (via food compost). After accounting for backyard food composting, 44% of sewer-connected households and 12% of septic-connected households report using a garbage disposal, which overall adds 2.9 kg C, 0.25 kgN, and 0.05 kg P hh^{-1} yr^{-1} to wastewater output fluxes. If garbage-disposal use were eliminated, we estimate that this would reduce nutrient loading to the Mississippi River (in WWTP effluent) by 0.17 kg C, 0.005 kg N, and

0.004 kg P hh^{-1} yr^{-1} (8.6%, 1.5%, and 1.9%, respectively, of loading to the Mississippi River), and to septic drainfields by 0.007 kg N and 0.001 kg P hh^{-1} yr^{-1} (0.4% and 0.5%, respectively, of loading to septic drainfields). Eliminating garbage-disposal use has no apparent tradeoffs or infrastructure needs, and ideally could also increase nutrient recycling if the food waste were composted instead.

Opportunity #2: Switch from septic to sewer. Another opportunity to reduce pollution is by switching septic-connected households to sewer. We estimate that an average 6% of N and 13% of P is removed from wastewater entering a household septic system, compared with 98% of N and 92% of P entering the Metro WWTP. Although only 12% of our study households are served by septic systems, total N and P fluxes to septic drainfields are considerably higher than fluxes in WWTP effluent to the Mississippi River: 582% for N, and 132% for P (Table 1.4). The higher removal efficiency of the WWTP would lead to a net decrease in pollution fluxes of 1.8 kg N hh^{-1} yr^{-1} and 0.26 kg P hh^{-1} yr^{-1}.

However, this approach has drawbacks that likely outweigh the benefits. Adding current septic-served households to the Metropolitan WWTP would increase pollution loading to the Mississippi River in a stretch that already exceeds ecoregion norms for C, N, and P (Minnesota Pollution Control Agency 2012). Loading would increase by 0.22 kg C, 0.04 kg N, and 0.03 kg P hh^{-1} yr^{-1}. Nutrient recycling (via land application of septage) would also decrease by 0.12 kg N and 0.04 kg P hh^{-1} yr^{-1} – 4% of current N recycling, and 10% of current P recycling. Expanding the sewershed would also incur substantial social and economic costs.

Opportunities to increase nutrient recycling to food production

Even if all nutrient pollution and "waste" were eliminated, there would still be a need to recycle household nutrients back into the food-production system to reduce reliance on mined P and manufactured N fertilizer. We view increasing nutrient recycling as a complement to reducing nutrient consumption, not a substitute. We explored waste-management opportunities that focus primarily on increasing P recycling to the food-production system, since P is non-substitutable in food production, with a secondary goal of

also increasing N recycling. While recycling C to the food-production system (as food-to-livestock, compost, or land-applied septage) is beneficial, we do not discuss these options quantitatively since the benefits per kg vary considerably depending on the specifics, and cannot be meaningfully compared. Recycled P and N, by contrast, directly replace fertilizer that would otherwise need to be mined or manufactured. There are likely minor differences in benefits per kg due to variable nutrient-use efficiencies (e.g. feeding livestock vs. applying compost to cropland) that we do not attempt to include in this study.

Opportunities to increase recycling can be broadly divided into two categories: source-separated recycling, where materials are separated out at each household (e.g. residential composting), or end-of-pipe recycling, where nutrients are recovered from waste streams at industrial scales (e.g. biosolids reuse). Source-separated recycling can allow individual households to take immediate action without waiting for societal-scale changes. In contrast, end-of-pipe recycling changes infrastructure rather than individual behaviors, which in essence guarantees 100% participation. Each approach offers different advantages for implementation, and increasing recycling is likely to involve a combination of both. Because of pathogen concerns, human feces and pet waste are generally not legal in source-separated recycling efforts such as composting. Food waste, yard waste, and even human urine can all be recycled as source-separated materials; we quantify each of these, as well as an end-of-pipe strategy to recover P from incinerator ash.

Our findings are encouraging: greatly increasing nutrient recycling could theoretically recycle more P and N to the food-production system than our households consume in human and pet food. Achieving that level of recycling is not easy, however; it would require infrastructure changes for ash P recovery and urine separation, plus 100% participation by households in source-separated recycling of food waste, yard waste, and urine.

Opportunity #1: Complete recycling of food waste. Potential increase: 0.36 kg P and 1.7 kg N hh^{-1} yr^{-1}. We estimated that 39% of food waste is currently recycled, either as food-to-livestock (from ICI sources) or backyard food compost (18% of households report composting food). The metro-area Solid Waste Management Coordinating Board estimates that there exists enough unused capacity to double current food-waste recycling, plus an additional

25% that is could be readily expanded with some investment (Solid Waste Management Coordinating Board 2007). If our study households (and food-preparation institutions) recycled all food waste that is currently disposed of in the garbage or via kitchen-sink garbage disposals, this would recycle an additional 0.36 kg P and 1.7 kg N hh^{-1} yr^{-1}, plus reduce water pollution as described above from eliminating garbage-disposal use.

Efforts to increase food composting are likely to combine backyard and off-site composting. We found considerable variation in backyard food-composting rates among municipalities and neighborhoods, which indicates flexibility in this behavior. Co-collection of food waste and yard waste in curbside organics collection has been successful in several other cities, as well as in a pilot project in a Minneapolis neighborhood that began in 2008 (Linden Hills Power and Light 2014). We recommend further studies to examine what factors drive differences in composting behavior, mindful that there may be a gap between actual and self-reported composting behavior, as there is for recycling. While it is also important to reduce spoilage and wastage of edible food within both the food supply chain and individual households (Parfitt et al. 2010), food waste consists of inedible portions (e.g. stems, rinds, peels, shells) which there will always be a need to compost.

Opportunity #2: Human urine separation and recycling. Potential increase: 0.98 kg P and 13.7 kg N hh^{-1} yr^{-1}. Human urine contains significant concentrations of P and N, and can be applied directly to cropland or processed into inorganic fertilizer forms such as struvite ($NH_4MgPO_4 \cdot 6H_2O$). The nutrient content of urine varies with the nutrient content and digestibility of the diet; we use data from Sweden to estimate that urine contains 67% of excreted P and 88% of excreted N (Vinneras 2002). Urine-separating toilets are becoming increasingly popular in Sweden, and a number of municipalities are developing large-scale urine-recycling programs to augment individual efforts of recycling urine to the household landscape or nearby farms (Vinneras2013). Urine storage tanks can serve single or multiple households, and are typically sized to require pumping only once per year, (~550 L/person, Vinneras 2002) which makes it possible for individual households to implement urine diversion without the need for large municipal projects or infrastructure. We estimate that complete adoption of urine separation and recycling has the potential to recycle an additional 0.98 kg P and 13.7 kg N hh^{-1} yr^{-1} to the food-production system.

Many large-scale visions for urine separation call for processing the N and P into inorganic fertilizers for easier storage and transport, in which case the synergistic benefits of urine separation are likely limited to the significant amounts of N that would also be recycled. Given the Metropolitan WWTP's current practice of incinerating biosolids, urine separation is the only option to recycle substantial amounts of N; without it, N recycling could only reach 44% of household food N fluxes. Widespread implementation would also incur large costs for new infrastructure to collect, store, process, and transport the urine, as well as social hurdles of changing toilet use.

Opportunity #3: Compost all household yard waste. Potential increase: 0.84 kg P and 6.3 kg N hh^{-1} yr^{-1}. Grass clippings and leaves offer an opportunity to harvest P and N from residential yards, including households where excess nutrients lead to potentially polluting landscape losses. This yard waste can be composted in county compost facilities and returned to the food-production system, either in residential and community gardens or cropland. Eighty-five percent of our study households (n=360) leave grass clippings on the lawn, and 42% of households either mow leaves into the lawn or compost leaves on-site (the HFC's landscape model does not distinguish among different methods of on-site leaf disposal). We used the HFC to model two landscape scenarios that increase the amount of grass clippings and raked leaves composted off-site (if a household only raked leaves from part of their property, we left this proportion unchanged as some properties are partly wooded).

In the first scenario, we maximized yard-waste composting by exporting all grass clippings and raked leaves to off-site compost. This scenario maximizes P harvest, even though it creates additional unmet ecosystem N demand (where N inputs are less than exports plus soil storage) and may require some additional N fertilization. We estimate that this would increase off-site compost fluxes by 0.84 kg P and 6.3 kg N hh^{-1} yr^{-1}. At the same time, we estimate that landscape losses would decrease by 0.01 kg P and 2.6 kg N hh^{-1} yr^{-1}. Unmet ecosystem N demand would increase by 0.83 kg N hh^{-1} yr^{-1}, increasing the percentage of households with unmet ecosystem N demand from 13% to 21%. Among households that would have an unmet N demand, its average magnitude would be 5.7 kg N hh^{-1} yr^{-1}. Some of the unmet demand may lead to increased N fertilizer use, which would offset the increase in N recycling; some may instead lead to "mining" of soil nutrients, reduce growth or

accumulation rates, or favor N-fixing species (e.g. clover) in the lawn. Our preliminary estimates indicate that N-fixation could meet only a fraction of the demand, roughly 0.7 kg N hh^{-1} yr^{-1} (S. E. Hobbie, *unpublished data*).

In the second scenario, we modeled an N-balanced yard-waste composting strategy by limiting yard-waste exports from each household to prevent unmet ecosystem N demand, even if that would reduce yard-waste exports below the household's current levels. Although our households' leaf litter has a higher N:P ratio than their grass clippings, meaning that an N-balanced P harvest would be maximized if grass clippings were preferentially composted, we assumed that people would preferentially leave grass clippings on the lawn (as 85% of households do now) instead of composting grass clippings off-site while mulching leaves into the lawn. If exporting all yard waste would create unmet ecosystem N demand, we modeled leaving only enough on the lawn (grass clippings first, then leaves if needed) to balance N demand. We estimate that this would increase off-site compost fluxes by 0.69 kg P hh^{-1} yr^{-1} and 5.1 kg N hh^{-1} yr^{-1}. Our calculations estimate that unmet ecosystem N demand would be reduced to zero under this scenario. Landscape losses of N would decrease by 5.7 kg N hh^{-1} yr^{-1} , the same as in the first scenario, but landscape losses of P would increase slightly by 0.006 kg P hh^{-1} yr^{-1}.

We interpret these estimates of increased composting as upper bounds for three reasons. First, some of the leaves currently composted on-site are likely already recycled into home food gardens, in which case composting them off-site would not increase nutrient recycling. Second, some compost is likely used for non-food uses or overapplied, as discussed above (see *Results: Current nutrient recycling*). Third, this rate of yard-waste removal may not be sustainable even if N were balanced. Total P fluxes in yard waste to compost would be 1.0 kg P hh^{-1} yr^{-1} under the first scenario and 0.85 kg P hh^{-1} yr^{-1} under the second, both substantially higher than inputs from pet waste (0.23 kg P hh^{-1} yr^{-1}) plus atmospheric deposition (0.03 kg P hh^{-1} yr^{-1}). Soils in our study area are P-rich, and weathering provides an additional P input to the household landscape, but to our knowledge this input has not been quantified. Before embarking on large-scale, long-term P harvests from residential landscapes, we recommend further study to estimate a sustainable yield rate.

Opportunity #4: Recover P from incinerator ash. Potential increase: 3.0 kg P and 0 kg N hh^{-1} yr^{-1}. When household garbage or WWTP solids are incinerated, C and N are burned off and all P is retained in the ash. Incineration destroys pathogens and organic pollutants, but heavy metals are retained in the ash at concentrations that are too high to allow direct land application of incinerator ash as a P fertilizer. While P recovery is not yet practiced on an industrial scale, methods have been demonstrated at the laboratory scale for separating a high percentage of P from the contaminating heavy metals (e.g. Franz2008, Mattenberger2008). If these or other methods can be scaled up, at some point the increasing cost of mining phosphate rock may make them economically attractive.

Currently, 3.0 kg P hh^{-1} yr^{-1} enter the landfill: 0.20 kg as unincinerated solid waste, 0.31 kg as solid-waste incinerator ash, and 2.5 kg as WWTP incinerator ash. Ramsey and Anoka Counties are both increasing the proportion of solid waste burned in waste-to-energy incinerators, with goals near 100% incineration by 2030 (Ramsey2012, Anoka2012). We therefore consider all 3.0 kg hh^{-1} yr^{-1} of landfilled P to be potentially recyclable through ash-recovery methods, either now or in the near future. In addition, incinerator ash is disposed of in designated ash landfills, which offers the possibility of using ash P recovery technology to mine these landfills for P.

Any changes that affect garbage, wastewater, or their transformations can also affect the amount of P in incinerator ash. Source-separated recycling of food waste or human urine would both reduce P fluxes to ash. Detergents and other products account for 41% of the P in incinerator ash; our estimate of this flux is highly uncertain (see above), and has decreased since our study due to the 2010 restrictions on P in automatic dishwasher detergent. There are also factors working to increase P fluxes to incinerator ash. The Metro WWTP has increased its P recovery efficiency since the 2008 data used in this study, and is working to increase it even further, (Metropolitan Council, *unpublished data*). Each percentage-point increase in removal efficiency redirects 0.03 kg P hh^{-1} yr^{-1} from the Mississippi River to incinerator ash. Public-health behavior-change campaigns are working to increase the proportion of dog feces that are scooped up and disposed of in the garbage; 100% scooping would increase pet-waste fluxes to incinerator ash by 0.12 kg P hh^{-1} yr^{-1}.

Implementing ash P recovery would yield immediate, large increases in recycled P. End-of-pipe P recycling, like ash P recovery or land-application of septage, is the only option to recycle P from materials that cannot currently be handled in source-separated recycling: pet waste, human feces, and detergents and other miscellaneous P-containing products entering wastewater or household garbage. Industrial-scale ash P recovery methods or alternative end-of-pipe approaches are therefore needed to close the cycle of this essential element. Furthermore, incinerator ash is disposed of in designated ash landfills, which makes it practical to mine these landfills for P using ash-recovery technology.

Combining recycling scenarios

These four recycling scenarios are not simply additive, since food waste and urine are currently part of WWTP ash. We estimated that combining all four recycling methods has the potential to recycle 4.3 kg P and 21.7 kg N hh^{-1} yr^{-1} if all households were to participate. This would bring total recycling fluxes up to 4.7 kg P and 24.6 kg N hh^{-1} yr^{-1} (there are no data on how urine separation might affect septage fluxes, so we left septage fluxes unchanged for this estimate).

Because household nutrients originate from a variety of sources, we also calculated what increases in recycling would be necessary for households to "close the cycle" and recycle as much P and N to the food production system as they consume in food. We excluded detergent fluxes from these calculations and focused primarily on the 2.5 kg P and 20.5 kg N hh^{-1} yr^{-1} in human and pet food (including both consumed and wasted food) that enters the household. We estimated that recycling all food waste, applying all septage to cropland (again leaving septage fluxes unchanged), and recovering all P from incinerator ash would recycle 2.0 kg P and 3.9 kg N hh^{-1} yr^{-1}, all of which originated from food (82% of P and 19% of N recycled). Adding an N-balanced yard-waste composting would harvest more nutrients from pet waste, as well as from other landscape sources such as atmospheric deposition or soil weathering, to increase recycling to 2.7 kg P and 9.0 kg N hh^{-1} yr^{-1} -- 109% and 44% of the P and N, respectively, in household food input fluxes. Even without urine separation or creating unmet ecosystem N demand, our study households could theoretically recycle more P to the food-production system than they consume in food. Adding urine separation, they

could do likewise for N, recycling 2.9 kg P and 22.7 kg N hh^{-1} yr^{-1} – 115% and 111% of P and N, respectively, in household food fluxes.

Ash P recovery is necessary for households to recycle more P than they consume in food; combining all other recycling opportunities would only recycle 1.5 kg P hh^{-1} yr^{-1} (59% of food fluxes) using an N-balanced yard-waste composting, or 1.6 kg P hh^{-1} yr^{-1} (65% of food fluxes) composting all yard waste. While ash recovery is essential to close the P cycle, we do not advocate sole reliance on ash P recovery because it is an energy-intensive process that, in contrast with source-separated recycling, does not recycle N or C, nor provide any other synergistic benefits. Food-to-livestock and composting both also return C, N, and micronutrients to the food-production system. Compost increases organic matter content in soils, which improves soil fertility. By reducing the amount of grain grown to feed livestock, food-to-livestock decreases agricultural pollution and inputs of land, energy, water, pesticides, and fertilizers. We recommend that food waste and yard waste continue to be recycled as source-separated materials, and their recycling increased to the maximum extent possible, not incinerated with household garbage to recycle their P alone.

Because we expect P scarcity to drive large-scale efforts to increase nutrient recycling, we expect that the costs of large-scale urine separation will not prove to be worth the small increases in overall P recycling (0.14 kg P hh^{-1} yr^{-1}) if ash P recovery were already in place. However, urine separation offers an opportunity for septic-connected households to substantially increase their P recycling. It could be a worthwhile return on investment for individual exurban households if nearby farmers were willing to purchase urine as a fertilizer, or if future septic regulations allow urine separation to replace more-expensive technologies to reduce N pollution of vulnerable aquifers in these areas. If ash recovery P technology is not implemented, though, urine separation could play a key role in nutrient recycling for all households.

For yard-waste composting, we favor an N-balanced approach to avoid creating excessive unmet ecosystem demand or increasing N fertilization. Even though a complete removal of yard waste could recycle more P, we estimate that an N-balanced approach could harvest more than enough P from residential landscapes to replace the food-derived P that cannot

currently be recycled (e.g. WWTP effluent, dog urine, and various small losses and inefficiencies). Although the precise N balance we modeled is likely unrealistic, we expect that a reasonable approximation can be achieved. Over 10% of Twin Cities households are willing to pay for a professional environmental assessment to obtain detailed yard-care recommendations and sustainability certification (C. Buyarski, *unpublished data*), and we recommend developing simplified guidelines that help other households use basic information (e.g. soil type, fertilizer use, watering practices) to estimate how much yard waste they can compost to increase nutrient recycling without incurring an N deficit.

Nutrient ratios

For N and P recycling to reduce reliance on manufactured and mined fertilizer, however, they must enter the food-production system in amounts and ratios that allow the nutrients to be used efficiently rather than wasted. For compost, septage, or other material applied directly to cropland, N or P applied in excess of crop requirements is likely to be lost. However, current practices do not generally consider both nutrients. For example, septage application to cropland is regulated to avoid overfertilization leading to N losses and pollution, but the regulations only require measuring and balancing N inputs with crop N needs, and do not regulate P (US Environmental Protection Agency 1994).

We compared the N:P ratios of our potential increases in recycling (except ash P recovery, which produces inorganic P fertilizer) with the N:P ratios of crop nutrient requirements for a food crop, corn (Minnesota's dominant crop; Ye 2011), and a forage crop, alfalfa hay. Nutrient requirements for corn have an N:P ratio of 5.2, whereas for alfalfa hay the N:P ratio is 10.7 (USDA 2014). These are both higher than the N:P ratio for septage of 2.8 (US Environmental Protection Agency 1994) or for food compost of 4.7 (Banks et al. 2011). Our estimates of potential yard waste compost, using an N-balanced approach, has an intermediate overall N:P ratio of 7.9, though we expect this to be heterogeneous depending on individual households' ratios of tree leaves to grass clippings. Finally, we estimated that urine has a higher N:P ratio than either crop's nutrient requirements: 14.0.

If current practices continue of regulating application based solely on N, we expect that much of the P returned to cropland would be applied in excess of crop requirements and

therefore wasted. When applying material with a lower N:P ratio than crop requirements, applying enough to meet crop N requirements would apply P in excess of crop requirements. For septage, which has the lowest N:P ratio, this would apply 1.9 times as much P as corn requires, or 3.6 times as much P as alfalfa hay requires. Even when applying smaller amounts of recycled material, or applying material such as urine with a higher N:P ratio, the P will still be wasted unless fertilizer P application is accurately reduced to account for these inputs. In order to reduce the need for mined P fertilizer, it is essential that nutrient recycling explicitly measure and regulate P application, not only N. Otherwise, recycling efforts will end up wasting much of the P they are attempting to conserve.

Conclusion

Households have the potential to recycle more P and N to the food-production system than they consume in food, a significant milestone in transitioning towards nutrient sustainability, but getting there requires substantial changes in how household wastes are disposed of and managed. Achieving this milestone for P recycling would require changing both individual household behaviors (greatly increasing source-separated recycling of food waste and yard waste) and large-scale societal infrastructure (recovering P from solid-waste and WWTP incinerator ash). In contrast, N recycling depends entirely on changes in household behaviors, and achieving this milestone must include urine separation (unless practices change at the Metropolitan WWTP to land-apply biosolids that are currently incinerated). Because our study demonstrates that household nutrient recycling can be increased ten-fold, we recommend focused studies to understand the drivers of recycling behaviors and design efforts to help households maximize their nutrient recycling. Furthermore, we urge substantial investments of financial, political, and social capital to help make both individual and infrastructure changes swift and widespread.

Improving waste management is not a panacea for nutrient sustainability, however. It does not reduce direct household consumption of fossil fuels, and our findings demonstrate that it can yield only minor reductions in nutrient pollution. Reducing household consumption of C, N, and P is the only way to make substantial improvements in these areas. We reinforce the findings of earlier TCHEP studies identifying the need to reduce

driving, flying, household energy use, and landscape fertilization, and we add additional strategies for focused efforts to further reduce nutrient pollution. In particular, we emphasize the importance of spatial considerations when working to reduce local air pollution, terrestrial N deposition, and water pollution. Small fluxes of nutrient pollution can still negatively and substantially impact ecosystem functions or human health, depending on the location where these fluxes occur. Differences in local population density are a greater driver of potential human exposure to volatile and particulate wood smoke C pollutants and home-heating NO_x-N, for example, than are differences in the nutrient fluxes themselves.

We recommend care when generalizing our specific results and recommendations, as nutrient management practices, social and biophysical context, and opportunities for change very considerably among cities (Metson et al., *in prep.*). Cities that do not incinerate solid waste and WWTP sludge, for example, would not benefit from ash P recovery, and as a result may not be able to achieve as high a degree of P recycling. However, we expect other cities with similar practices, but for which similarly detailed analyses have not yet been conducted, can also benefit from the insights in this study. We contribute this work to the growing body of nutrient-sustainability research to guide transitions towards nutrient sustainability not only in the Twin Cities, but also in other urban areas working to address the challenges of nutrient pollution and nutrient scarcity.

Table 1.1: Revised annual per-capita food waste fluxes for residential and industrial/commercial/institutional (ICI) sources. Residential food waste measured at the landfill was from all households, with and without garbage disposals; since the former landfilled only a portion of their food waste, this did not measure all residential food waste. We therefore calculated total residential food waste (TRFW) as described for the original HFC: Residential food waste measured at landfill = (0.48 x TRFW) + (0.52 x (TRFW – garbage-disposal fluxes)), where 52% of our study households reported using a garbage disposal (Fissore et al. 2011, Appendix A). Siegrist and coauthors (1976) reported C, N, and P fluxes directly; all others reported wet weight of food which we converted to element fluxes using stoichiometry from Banks and coauthors (2011), as detailed in the text.

	C	**N**	**P**
	kg capita $^{-1}$ yr $^{-1}$		
ICI to landfill (Beck 2000)	4.12	0.30	0.06
ICI to livestock (SWCMB 2007)	4.69	0.34	0.07
Total ICI food waste	**8.81**	**0.64**	**0.14**
Total residential food waste	**6.24**	**0.47**	**0.10**
Total food waste (ICI + Residential)	**15.05**	**1.12**	**0.24**
values for calculating total residential food waste (see caption):			
Residential to garbage disposal (Siegrist et al. 1976) (only for households using a garbage disposal)	2.67	0.23	0.05
Residential measured at landfill from all households (Beck 2000)	4.85	0.35	0.05

Table 1.2: Coefficients for CH_4, N_2O, and NO_x emissions. Original sources report these values per unit heat content or fuel volume. We converted these to mass ratios per kg CO_2-C, using either reported CO_2 emissions or HFC calculations of C content (for wood and lawnmower fuel). Vehicle emissions are not listed here, as they are reported per mile driven and therefore cannot be converted to a constant mass ratio since the HFC incorporates model-specific fuel efficiency data into CO_2 calculations. Vehicle CH_4 and N_2O fluxes vary based on model year and car/truck designation using data from the EPA (2008); vehicle NOx calculations are described in the original HFC.

Combustion Source	CH_4-C	N_2O-N	NO_x-N
	kg per kg CO_2-C		
Natural gas	$2.5x10^{-4}$ (a)	$4.4x10^{-6}$ (a)	$8.6x10^{-4}$ (b)
Propane	$4.4x10^{-4}$ (a)	$2.2x10^{-5}$ (a)	$1.3x10^{-3}$ (c)
Oil	$3.7x10^{-4}$ (a)	$1.9x10^{-5}$ (a)	$8.9x10^{-4}$ (c)
Wood heating	$7.8x10^{-3}$ (a)	$8.8x10^{-5}$ (a)	$9.3x10^{-4}$ (b)
Electricity	$2.8x10^{-5}$ (d)	$3.8x10^{-5}$ (d)	$2.1x10^{-3}$ (e)
Air travel	$7.7x10^{-5}$ (a)	$7.5x10^{-5}$ (a)	$3.1x10^{-3}$ (domestic) (g) $6.1x10^{-3}$ (international) (g)
Lawnmower	$3.1x10^{-3}$ (f)	$3.9x10^{-4}$ (f)	$1.6x10^{-3}$ (f)

Sources:
(a) US Energy Information Authority 2011
(b) US Environmental Protection Agency 1995
(c) Office of Energy Efficiency and Renewable Energy 2011
(d) US Energy Information Authority 2002
(e) US Energy Information Authority 2013
(f) Christensen 2011
(g) Fissore and coauthors (2011, Appendix A) using data from multiple sources

Table 1.3: Household output fluxes of (a) carbon, (b) nitrogen, and (c) phosphorus (kg hh^{-1} yr^{-1}) by HFC component and form. Summary statistics are the mean, median, first and third quartiles (Q1 and Q3), and two measures of dispersion: the median absolute deviation (MAD) and the Gini coefficient (described in text). *n* = the number of households which had data to calculate the flux. Zero values indicate that the estimated value was <0.005. Only the mean is reported for lawnmower fluxes since we assumed fixed per-household values.

(a) Carbon	**Mean**	**Q1**	**Median**	**Q3**	**MAD**	**Gini**	***n***
Air and car travel							
Fossil-fuel CO_2	3501.1	1619.2	3005.7	4539.7	2140.2	0.40	2763
CH_4	0.52	0.20	0.39	0.64	0.31	0.48	2763
Household energy							
Fossil-fuel CO_2	3232.7	2474.9	3052.7	3789.5	941.8	0.19	1843
Non-fossil-fuel CO_2	63.8	0	0	0	0	0.97	
Wood smoke	1.6	0	0	0	0	0.97	2763
CH_4	0.94	0.35	0.42	0.53	0.13	0.60	1843
Food waste							
Garbage	20.4	10.4	16.5	24.7	9.1	0.31	2763
Livestock	11.9	9.4	9.4	14.1	7.0	0.26	2763
Food compost	3.0	0	0	0	0	0.86	2763
Wastewater	2.9	0	0	5.3	0	0.70	2763
Paper & plastic							
Garbage	143.9	110.7	110.7	184.7	82.1	0.27	2763
Recycling	64.6	29.1	61.9	84.1	38.5	0.39	2763
Pets							
Non-fossil-fuel CO_2	21.9	0	2.5	34.0	3.8	0.74	2763
Garbage	0.76	0	0.09	1.2	0.14	0.74	2763
Human diet							
Non-fossil-fuel CO_2	248.3	166.6	220.1	320.4	110.3	0.28	2763
Wastewater	6.8	4.4	5.9	8.8	3.1	0.29	2763
Detergents etc.							
Wastewater	26.5	20.9	20.9	31.4	15.5	0.26	2763
Household landscape							
Non-fossil-fuel CO_2	382.1	93.9	175.6	292.0	137.5	0.63	360

Storage in wood	283.6	55.8	110.8	204.6	94.0	0.69	360
Yard waste county compost	60.4	0	22.0	56.6	32.6	0.76	360
Storage in soil	38.9	0	18.3	37.0	27.1	0.70	360
Lawnmower fossil-fuel CO_2	10.1	n/a	n/a	n/a	n/a	n/a	n/a
Lawnmower CH_4	0.03	n/a	n/a	n/a	n/a	n/a	n/a

(b) Nitrogen	Mean	Q1	Median	Q3	MAD	Gini	*n*
Air and car travel							
NO_x	13.7	6.2	11.5	17.5	8.3	0.42	2763
N_2O	0.50	0.14	0.33	0.72	0.36	0.52	2763
Household energy							
NO_x	4.9	3.5	4.5	5.8	1.7	0.22	1843
N_2O	0.08	0.05	0.07	0.09	0.03	0.29	1843
Food waste							
Garbage	1.5	0.77	1.2	1.8	0.64	0.31	2763
Livestock	0.86	0.68	0.68	1.0	0.50	0.26	2763
Wastewater	0.25	0	0	0.46	0	0.70	2763
Food compost	0.23	0	0	0.00	0	0.86	2763
Pets							
Garbage	0.36	0	0.13	0.56	0.19	0.68	2763
Human diet							
Wastewater	15.6	10.4	13.8	20.0	6.9	0.28	2763
Detergents etc.							
Wastewater	1.8	1.4	1.4	2.1	1.0	0.26	2763
Household landscape							
Landscape losses	8.4	1.4	4.2	9.2	5.4	0.63	360
Storage in soil	3.2	0	1.5	3.1	2.3	0.70	360
Yard waste county compost	1.6	0	0.53	1.6	0.79	0.78	360
Storage in wood	1.6	0.32	0.63	1.2	0.53	0.69	360
Lawnmower NO_x	0.02	n/a	n/a	n/a	n/a	n/a	n/a
Lawnmower N_2O	0	n/a	n/a	n/a	n/a	n/a	n/a

(c) Phosphorus	Mean	Q1	Median	Q3	MAD	Gini	*n*
Food waste							
Garbage	0.30	0.16	0.24	0.36	0.12	0.31	2763
Livestock	0.18	0.14	0.14	0.21	0.10	0.26	2763
Wastewater	0.05	0	0.00	0.10	0.00	0.70	2763
Food compost	0.05	0	0.00	0.00	0.00	0.86	2763
Pets							
Garbage	0.21	0.00	0.05	0.32	0.07	0.71	2763
Human Diet							
Wastewater	1.5	0.98	1.3	1.9	0.65	0.28	2763
Detergents etc.							
Wastewater	1.5	1.2	1.2	1.8	0.89	0.26	2763
Household landscape							
Storage in soil	0.49	0.00	0.23	0.46	0.34	0.70	360
Storage in wood	0.18	0.04	0.07	0.13	0.06	0.69	360
Yard waste county compost	0.16	0.00	0.04	0.13	0.06	0.81	360
Landscape losses	0.09	0.00	0.00	0.00	0.00	0.90	360

Table 1.4: Eventual fates of household C, N, and P fluxes, grouped into pollutants, inert forms, and recycling.

	C		**N**		**P**	
	kg hh^{-1} yr^{-1}	%	*kg hh^{-1} yr^{-1}*	%	*kg hh^{-1} yr^{-1}*	%
Pollution	**6748.9**	**83**	**29.7**	**54**	**0.59**	**13**
Fossil-fuel CO_2	6743.9	83	----		----	
CH_4	1.5	<0.1	----		----	
N_2O	----		0.58	1.1	----	
NO_x	----		18.6	34	----	
Wood smoke	1.6	<0.1	----		----	
Septic drainfield	----		1.8	3.3	0.29	6.1
Mississippi River	1.9	<0.1	0.31	0.6	0.22	4.6
Landscape losses	----		8.4	15	0.09	1.9
Inert Forms	**1267.3**	**16**	**22.0**	**40**	**3.7**	**78**
Non-fossil-fuel CO_2	877.4	11	----		----	
N_2	----		16.5	30	----	
Landfill	66.0	0.8	0.73	1.3	3.0	64
Septic drainfield	1.4	<0.1	----		----	
Storage in wood	283.6	3.5	1.6	2.9	0.18	3.8
Storage in soil	38.9	0.5	3.2	5.9	0.49	10
Recycled	**110.4**	**1.4**	**2.83**	**5.2**	**0.43**	**9.2**
Paper/plastic recycling	64.6	0.8	----		----	
Land application	34.0	0.4	1.97	3.6	0.25	5.4
Livestock	11.9	0.1	0.86	1.6	0.18	3.8

Table 1.5: Average household global warming potential, in kg CO_2-C equivalent hh^{-1} yr^{-1}. Totals are summed down by gas and across by activity, plus a grand total for all household GWP.

	CO_2	CH_4	N_2O	*Total*
Air travel	623.2	0.44	6.0	*629.6*
Car travel	2902.7	4.0	56.0	*2962.7*
Home energy	3232.7	8.5	9.8	*3251.1*
Lawnmower	10.1	0.28	0.50	*10.9*
Total	*6768.7*	*13.3*	*72.3*	**6854.3**

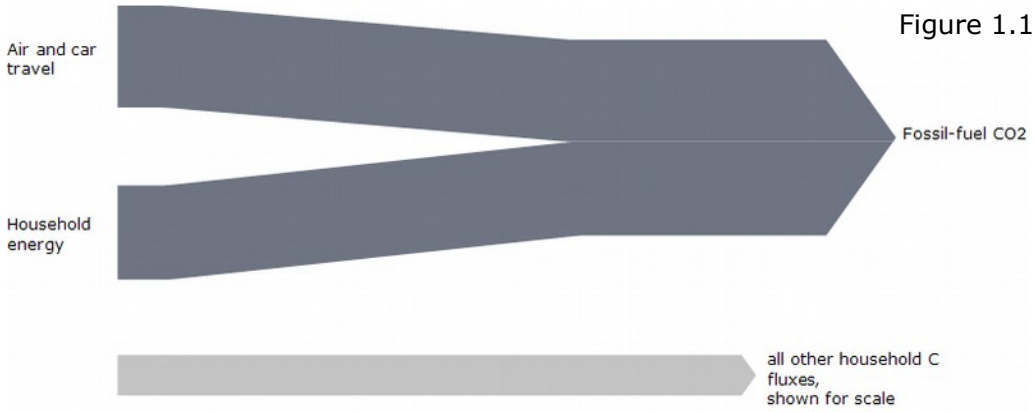


Figure 1.1: Carbon

(a)

(b)

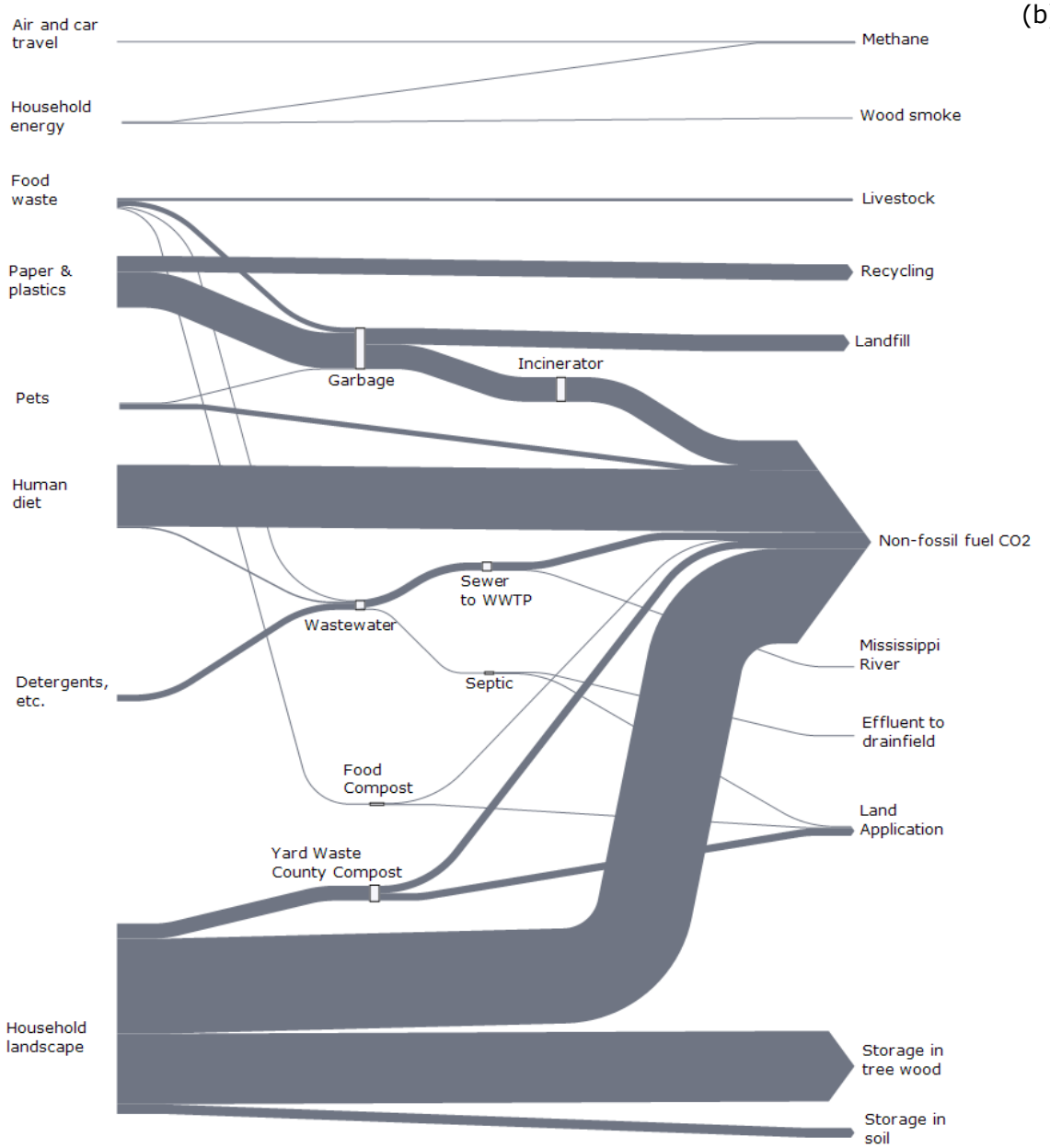


Figure 1.1: Carbon fluxes, where the width of the arrow is proportional to the size of the flux (kg C hh^{-1} yr^{-1}). Small flux arrows are not proportional, since they are drawn with a minimum width of one pixel to remain visible. Figure 1a focuses on fossil-fuel CO_2 fluxes, which make up 83% of household C fluxes. All other fluxes are shown combined into a single flux arrow, for comparison. Figure 1b omits fossil-fuel CO_2 fluxes to show the details of other C fluxes. Lawnmower fluxes are not drawn; they are one or more orders of magnitude smaller than other fluxes in the same figure, and because of the minimum arrow width give a misleading impression of being much larger than they actually are.

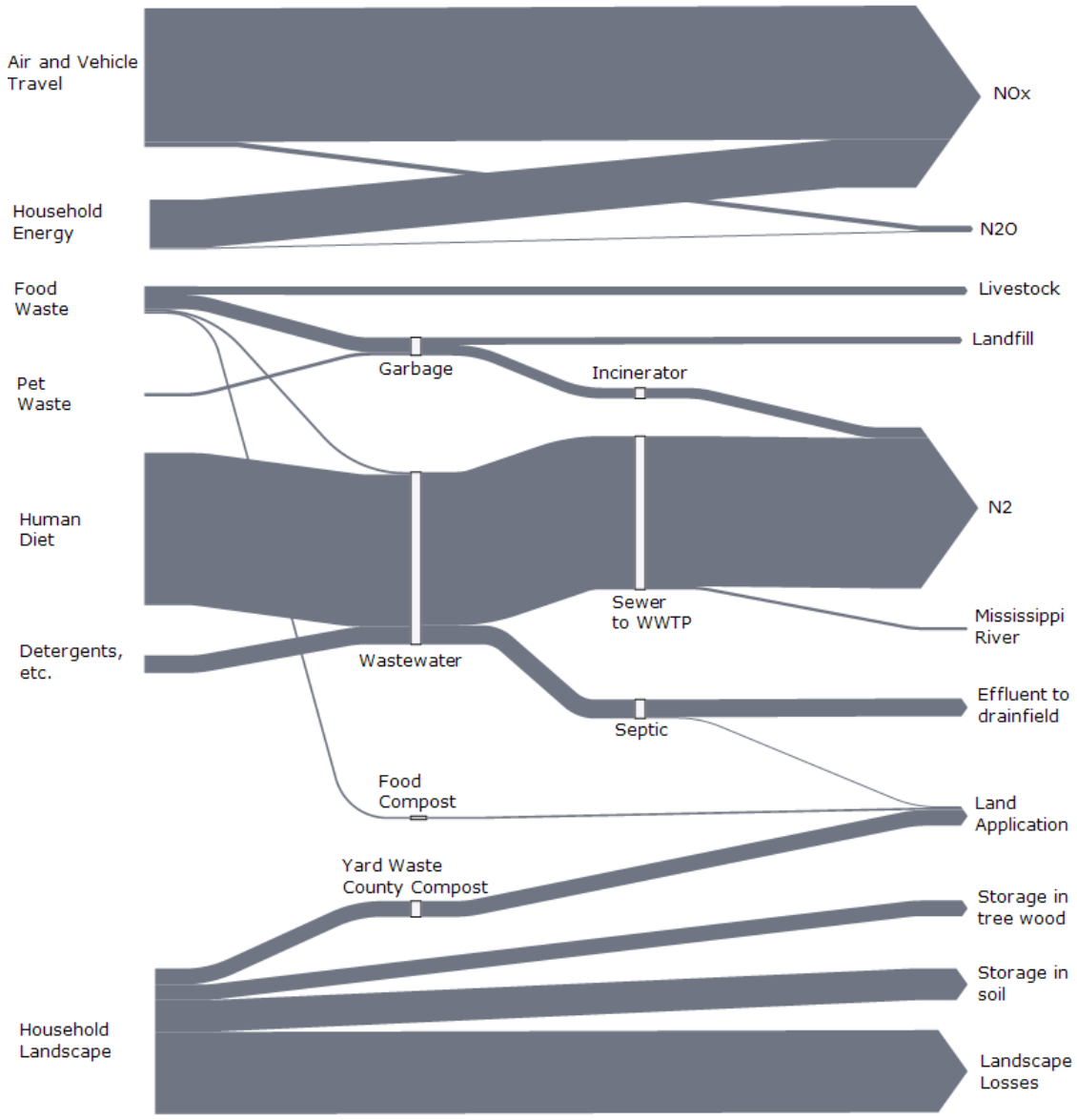


Figure 1.2: Nitrogen fluxes, where the width of the arrow is proportional to the size of the flux (kg N hh^{-1} y^{r-1}). Small flux arrows are not proportional, since they are drawn with a minimum width of one pixel to remain visible. Lawnmower fluxes are not drawn; they are one or more orders of magnitude smaller than other fluxes in the figure, and because of the minimum arrow width give a misleading impression of being much larger than they actually are.

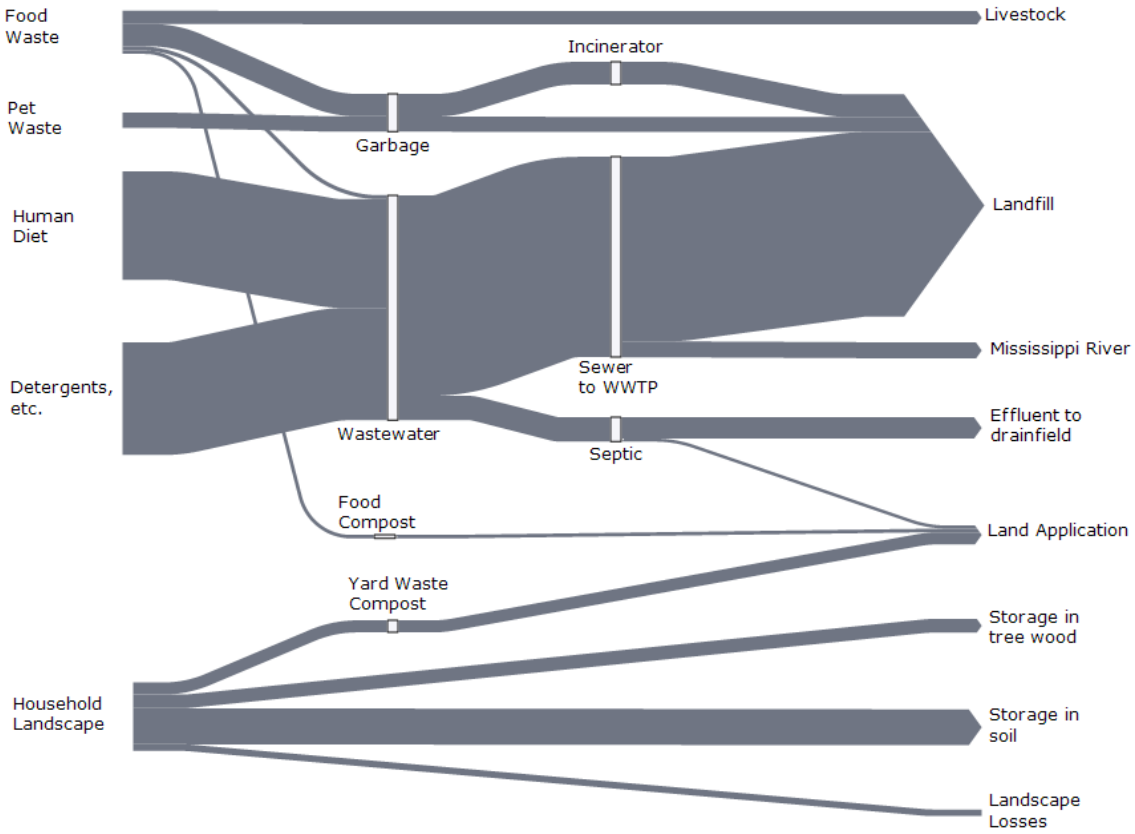


Figure 1.3: Phosphorus fluxes, where the width of the arrow is proportional to the size of the flux (kg P hh^{-1} y^{r-1}). Small flux arrows are not proportional, since they are drawn with a minimum width of one pixel to remain visible.

Chapter 2:

Can urban trees reduce nutrient leaching to groundwater?

Abstract

Many urban waterways suffer from excess nitrogen (N) and phosphorus (P) feeding algal blooms, which cause lower water clarity and oxygen levels, bad odor and taste, and the loss of desirable species. Nutrient movement from land to water is likely to be influenced by urban vegetation, but there are few empirical studies addressing this. In this study, we examined whether or not urban trees can reduce nutrient leaching to groundwater, an important pathway that has received less attention than stormwater. We sampled thirty-three trees of fourteen species, and seven open grassy areas, across three city parks in Saint Paul, Minnesota. We installed lysimeters at 60cm depth to collect soil water and measure nutrient concentrations approximately biweekly from July 2011 through October 2013, and modeled water fluxes using the BROOK90 hydrologic model. We also measured soil nutrient pools (bulk C and N, KCl-extractable inorganic N, Brays-P), tree tissue nutrient concentrations (C, N, and P of green leaves, leaf litter, and roots), and tree size parameters (leaf biomass, leaf area index) to explore correlations with nutrient leaching. Trees had similar or lower N leaching than turfgrass in 2012 but higher N leaching in 2013; trees reduced P leaching compared with turfgrass in both 2012 and 2013, with lower leaching under deciduous trees than evergreens. Scaling up our measurements to the Capitol Region Watershed (~17,400 ha), we estimated that trees reduced P leaching to groundwater by 533 kg in 2012 and 1201 kg in 2013. Removing the same amounts of P with stormwater infrastructure would cost $2.2 million and $5.0 million per year, respectively.

Introduction

Urban trees enhance human well-being in many ways, from improving air quality (Nowak et al. 2006b) to reducing crime rates (Kuo and Sullivan 2001), but less is understood about how urban trees affect the water quality of local lakes and streams. Because trees'

physiological traits contrast sharply with those of the turfgrasses that dominate residential landscapes, they are likely to alter nutrient cycling and the movement of nitrogen (N) and phosphorus (P) from land to water. Yet there has been little research on how trees or other urban vegetation affect nutrient transport to stormwater and groundwater, which in turn flow to local lakes and streams. Although most urban water-quality efforts focus on stormwater, urban groundwater has been shown to contain concentrations of N and P high enough to contribute to water-quality problems (Trojan et al. 2003, Janke et al. 2013). Excess nutrients drive algal blooms that cause lower water clarity and oxygen levels, bad odor and taste, and the loss of desirable species. Urban trees and other "green infrastructure" may be able to reduce nutrient pollution and play an important role in improving the water quality and ecosystem service provisioning of local lakes and streams.

Many parts of American cities and towns outside the dense downtown cores, such as residential neighborhoods, could be described as "urban savannas" with isolated, open-grown trees and a grassy understory. As in natural savannas, trees in urban savannas have been shown to create altered microclimates with characteristics such as cooler soil and ground-surface temperatures (Peters and McFadden 2010) and lower wind velocities (Dewalle and Heisler 2012); their effects on air temperature, relative humidity, and soil moisture are less clear (Miller 1997). Trees differ from turfgrasses in ways that are also likely to affect nutrient cycling; for example, they have deeper roots with greater lateral spread, strong seasonal patterns of litterfall, and woodier tissues. Differences between trees and grasses are well documented in tropical savannas, where a consistent picture has emerged of trees creating "fertile islands" with higher soil carbon, nitrogen, and phosphorus in the sub-canopy soil than in open grassy areas, even in savannas where the trees are not symbiotic N-fixers (see reviews in Vetaas 1992, Rhoades 1997). In many tropical farmlands, trees are traditionally retained in pastures and croplands, in part because of beneficial effects on soil fertility and crop yields (Rhoades 1997).

In temperate savannas and grasslands, however, this pattern is less clear. These systems consistently exhibit significant differences between nutrient cycling under trees and grasses, but the magnitude and even the direction of trees' influence vary. In some sites, for example, trees increased N mineralization (Dijkstra et al. 2006) and soil organic matter (Gill and Burke

1999), but in other sites trees decreased them (Jackson et al. 2002, McCarron et al. 2003). Understanding these idiosyncratic results is complicated by two main factors: First, many natural savannas and tree-invaded grasslands have only one or two tree species, preventing cross-species comparisons to disentangle the effects of species and sites. Second, trees in savannas and grasslands do not establish randomly, but establish preferentially in microsites with favorable soil properties (e.g. texture, pH, base saturation; Geiger et al. 1994). Differences in nutrient cycling beneath trees are influenced by pre-existing abiotic differences as well as the effects of the trees themselves. Thus, while results from natural systems support the idea that urban trees are likely to be important drivers of nutrient cycling, they offer no clear consensus on the magnitude or direction of these effects.

A handful of studies have investigated nutrient leaching in urban systems, also with contradictory results. For example, in newly-planted experimental plots, a mix of woody and herbaceous ornamental species had higher N and P leaching fluxes than turfgrass plots, even though turfgrass plots were more heavily fertilized (Loper et al. 2013). The authors caution that these results may change once the ornamental species are fully established, however, since their data only span the first year after planting. Another set of experimental plots, where sampling began 1.5 years after planting, found opposite results (Qin et al. 2013): plots with 10% tree cover had higher N leaching than those with 25% or 40% tree+shrub cover, and P leaching was similar across all plots, even though trees and shrubs were fertilized more heavily than turfgrass. Another study that compared nitrate leaching below several different categories of established landscaping at a horticultural garden in Rhode Island (Amador et al. 2007) did not find differences in total NO_3-N fluxes among turfgrass, evergreen trees, and deciduous trees, even though turfgrass areas were fertilized and trees were not. Sample sizes were limited, however, and the trees were mulched annually with pine bark, which may have obscured species effects on nitrogen cycling and leaching. Other studies comparing nutrient leaching under trees (and/or shrubs) vs. turfgrass found higher nutrient leaching under vegetation types that received more fertilizer (e.g. Erickson et al. 2005, 2008, Groffman et al. 2009), so vegetation effects could not be separated from fertilization effects in these studies.

In managing urban forests to improve water quality, differences among tree species are also important. The expansion and turnover of urban forests present a large-scale

opportunity to shift tree species composition in favor of species that improve water quality, but current scientific knowledge is not sufficient to guide these decisions. Data from non-urban forests support the idea that nutrient leaching is likely to differ significantly among tree species, and to be related to plant physiological traits. Nitrate leaching was inversely related to forest floor (Oe + Oa layers) C:N ratio in European coniferous forests (Gundersen et al. 1998) and Northeastern USA hardwood forests (Lovett et al. 2002). These are all mixed species forests, not monocultures, so the authors used forest floor C:N ratio as a way of combining the different species' traits into a single variable. The Lovett study in particular compared sites that were chosen to have very similar abiotic conditions in order to focus on biotic effects.

In this study, we examined nutrient leaching under multiple urban tree species in the same environmental conditions and management practices, and correlated nutrient leaching with tree physiological traits. In addition to understanding how plant traits control nutrient movement from land to water, our results can help homeowners, city foresters, and other land managers select tree species that reduce nutrient pollution.

We measured N and P leaching under 14 common species of urban trees, as well as open turfgrass areas. Our specific objectives were to: 1) compare nutrient leaching between trees and turfgrass, 2) compare nutrient leaching among tree species, and relate those differences to plant physiological traits, and 3) scale up our leaching estimates to estimate the watershed-scale effects of trees on nutrient leaching. For our first objective, we hypothesized that trees would have lower leaching of N and P to groundwater than turfgrass. We expected that trees take up more N and P than turfgrass, and also provide soil microbes with more organic carbon than turfgrass, thereby increasing nutrient immobilization rates. Unlike prairie grasses that have very deep root systems, turfgrasses are more shallow-rooted and do not build deep organic-rich soils (Jackson et al. 1996).

For our second objective, we hypothesized that tree root and leaf-litter nutrient concentrations would be positively correlated with nutrient leaching. We expected that tree species with lower nutrient concentration would produce more recalcitrant litter with higher N and P immobilization rates, thus decreasing the amount of nutrients in readily leachable forms. We also hypothesized that evergreens would reduce nutrient leaching over a longer

portion of the growing season than deciduous trees, as they are photosynthesizing, transpiring, and taking up nutrients during early spring and late fall when the soil is thawed but deciduous trees are not in leaf. During those periods, we expected lower N and P leaching under evergreen than deciduous trees, especially during the fall when litterfall adds a pulse of nutrients to the soil.

For our third objective, we scaled up our results to the Capitol Region Watershed, an area of approximately 17,400 ha draining into the Mississippi River and encompassing portions of the cities of Falcon Heights, Lauderdale, Maplewood, Roseville and Saint Paul. We hypothesized that even small differences among vegetation types in N or P leaching per m^2 could result in watershed-scale effects large enough to substantially alter water quality.

Methods

We measured nutrient leaching beneath urban trees and open turfgrass areas in Saint Paul, Minnesota, USA. We used tension lysimeters to obtain repeated samples of soil water to measure nutrient concentrations, and combined these data with modeled estimates of vertical soil water flux to calculate nutrient leaching fluxes. We also measured soil nutrient pools (bulk C and N, KCl-extractable inorganic N, Brays-P), tree tissue nutrient concentrations (C, N, and P of green leaves, leaf litter, and roots), and tree size parameters (leaf biomass, leaf area index) to explore correlations with nutrient leaching.

Study sites and sampling locations

Our study sites were three city parks (~1.6 ha each) in residential neighborhoods of western Saint Paul, Minnesota, USA, roughly in a NW/SE line spanning 6 km.. Saint Paul has a humid continental climate, with average annual precipitation of 780mm (140mm as snow) and average daily high and low temperatures ranging from -4.6 and -13.6°C in January to 28.6 and 17.8°C in July (1981-2010 averages for the Minneapolis-Saint Paul International Airport weather station, compiled by NOAA). Groundwater is fairly shallow, generally <10m below the surface.

All three parks receive identical minimal management: they are not fertilized nor irrigated, and leaf litter and grass clippings are mulched into the grass with a mulching

mower. Soils are derived from glacial till and classified as sandy to silty loams overlying coarse sand (USDA Web Soil Survey). During lysimeter installation, we observed that the depth to the coarse sand varied considerably even within a park, generally ranging from 30 to 75 cm.

Across these parks, we selected forty sampling sites, 33 under trees (23 deciduous, 10 evergreen) and 7 in open turfgrass areas (at least one tree-height away from all trees to minimize tree influences). Focal trees were healthy, mature trees in relatively flat areas, without impervious surfaces or other management activities (e.g. mulch, non-turfgrass plantings) under the canopy. Focal tree species were selected to represent common deciduous and evergreen urban tree species with a range of leaf N and P concentrations: *Abies concolor* (white/concolor fir, n=2), *Acer platanoides* (Norway maple, n=4), *A. saccharinum* (silver maple, n=2), *Celtis occidentalis* (hackberry, n=5), *Fraxinus pennsylvanica* (green ash, n=5), *Ginkgo biloba* (ginkgo, a deciduous broad-leaved gymnosperm, n=1), *Gleditsia triacanthos* (honeylocust, a symbiotic N-fixer, n=1), *Pinus resinosa* (red pine, n=1), *P. strobus* (Eastern white pine, n=2), *P. sylvestris* (Scots pine, n=3), *P. ponderosa* (Ponderosa pine, n=2), *Quercus macrocarpa* (bur oak, n=3), *Tilia cordata* (little-leaf linden, n=2). Two of our focal *F. pennsylvanica* trees were removed by Saint Paul Parks & Recreation over the 2011-2012 winter (cut down and stumps ground below the soil surface) to prevent them from becoming hosts for emerald ash borer. We continued to sample the lysimeters from the sites of these removed trees to help understand if any tree-caused differences would be transient or persistent.

Lysimeters

At each focal tree or open turfgrass area, we installed one tension lysimeter at 60 cm depth, below 80-90% of tree roots (Crow 2005), during early summer 2011. (We received formal permission from the City of Saint Paul before beginning sampling or equipment installation.) For trees, we located our lysimeters along a random compass bearing from the focal tree's trunk, halfway between the trunk and the edge of the tree canopy (dripline). We excluded locations that were within one tree-height of a neighboring heterospecific tree unless it was on the opposite side of the focal tree. For turfgrass areas, we started at a

location near the center of the open area and then placed the lysimeter 2m away along a random compass bearing. Lysimeters were constructed of a porous ceramic cup (SoilMoisture Equipment Corp, Santa Barbara, CA) attached to a 30cm-long segment of PVC pipe, and capped with a rubber stopper. A flexible, inert plastic tubing line led to the surface for sampling. Lysimeters were flushed with 10% HCl and several rinses of nanopure water after assembly, and after installation we collected and discarded 2-3 samples of soil water to further flush and equilibrate them. To install lysimeters, we augured an 8cm-diameter vertical borehole, seated the lysimeter in silica flour, backfilled the borehole with clean sand, and sealed the top of the borehole with a layer of bentonite clay to prevent preferential water flow down the borehole.

We sampled each lysimeter approximately biweekly during the growing season (thaw to snow cover) unless the soil was too dry to yield a sample, from July 2011 through October 2013. To sample a lysimeter, we pumped a -50kPa vacuum, sealed the lysimeter, and returned 1-2d later to pump out the water it had collected. Samples were transported back to lab and frozen until nutrient analyses. Additional filtration was not required because the soil water had already passed through the porous ceramic cup of the lysimeter. We analyzed lysimeter samples for total dissolved N using a Shimadzu TOC Vcpn analyzer (Shimadzu Scientific Instruments, Columbia, MD), total dissolved P using a persulfate digest followed by molybdate-blue colorimetry, NO_3^--N using vanadium-oxidation colorimetry, and soluble reactive P by molybdate-blue chemistry. NH_4^+-N concentrations in soil water, measured using salicylate colorimetery, were too low to detect (< 0.1 mg/L).

Soil and tree measurements

During lysimeter installation, we collected borehole soil samples from 0-10cm, 10-20cm, 20-40cm, and 40-60cm depth horizons, and also measured the borehole volume (to calculate bulk density) at each depth by lining the borehole with a thin plastic bag and measuring the amount of water needed to fill it. We measured total soil C and N using a Variomax CN combustion analyzer (Elementar Analysensysteme, GmbH, Haunau, Germany). For measuring extractable nutrient pools, we collected additional 0-10cm soil samples in July 2011 at four points (on cardinal directions) around each tree, halfway between the trunk and

dripline, or at four points 2m from each turfgrass lysimeter. We composited these soil samples and extracted a 10g subsample with 2M KCl to measure NO_3^- and NH_4^+, and incubated a second 10g subsample in the dark at room temperature for 10d to determine potential rates of net mineralization and net nitrification. We then extracted the incubated subsample in the same manner, and calculated net N mineralization and nitrification as the difference between final and initial extractable inorganic N or NO_3^--N, respectively. We also extracted a third 5g subsample for Brays-P (Bray and Kurtz 1945).

We measured each tree's height, trunk diameter, canopy width, canopy fullness, etc. in summer 2011 and used the urban-specific software i-Tree Eco v5.0.8 (www.itreetools.org) to estimate growth rate (reported in as "gross carbon sequestration," which is the annual increase in kg C stored in tree wood) leaf biomass, canopy area, and leaf area index (LAI). We measured C, N, and P contents of fully expanded sun leaves (collected July 2011), fresh leaf litter (fall 2011), and fine, live roots (<2mm diameter, collected during lysimeter installation). We measured leaf %C and %N using a Variomax CN combustion analyzer (Elementar Analysensysteme, GmbH, Haunau, Germany), and litter and root %C and %N using a Costech ECS4010 element analyzer (Costech Analytical, Valencia, California, USA) at the University of Nebraska, Lincoln. P concentrations of all tissue samples were measured by ashing the sample (300ºC for 30 min followed by 550°C for 2 h) and extracting with 10N sulfuric acid, followed by molybdate-blue colorimetry (DeMott et al. 1998).

Hydrologic modeling

We modeled daily vertical water fluxes at 60 cm depth using the BROOK90 hydrologic model (Federer et al. 2003). For model inputs, we obtained daily precipitation and temperature data from the University of Minnesota climate station (located within 6 km of all study sites), as well as wind speed, dewpoint, and solar radiation data from the Saint Paul downtown airport (located within 12 km of all study sites). We modeled 2007-2010 as an initialization and calibration period for the model prior to our output period of 2011-2013. Using a turfgrass LAI of 1.5 (after Milesi et al. 2005) and default model parameters yielded good predictions for turfgrass evapotranspiration (ET) in 2008 when compared with eddy-flux measurements from a nearby study (Peters et al. 2011). For trees, we reduced the

maximum leaf conductance parameter so modeled tree transpiration in 2008 matched values calculated from 2008 sapflow measurements on similar open-grown urban trees in parklike conditions, within 10 km of our study sites (Peters et al. 2010). Peters and coauthors found that open-grown urban trees had lower ET per m^2 projected canopy area than open, sunny turfgrass that was not irrigated or fertilized.

Because LAI exerts strong controls on ET, and hence soil water fluxes, we modeled each focal tree separately using that tree's LAI and height, with BROOK90 set to calculate ET from a sparse canopy. We modeled all evergreen trees without a turfgrass understory, even though some of our focal evergreens had some grass beneath the canopy. All of our focal deciduous trees had a turfgrass understory. To model this in BROOK90, which cannot explicitly model two separate canopy layers, we increased root density to the sum of the model's default deciduous and turfgrass root densities, added the turfgrass LAI (1.5, constant across the whole year) to the tree's LAI (which increased from zero to maximum in April, and decreased from maximum to zero in November), and reduced the light-extinction parameter to simulate partly-shaded turfgrass.

Flux calculations

We created a daily timeseries of estimated soil water N and P concentrations by interpolating linearly between soil water samples for each lysimeter. We interpolated across any missing values within a calendar year's sampling period, but we did not extrapolate beyond the first and last sample collected for that tree or turfgrass area. We multiplied each day's concentration estimates (in mg/L or μg/L) by modeled soil water fluxes (in mm/d, equivalent to L m^{-2} d^{-1}) to yield vertical soil water fluxes, which we express in mg m^{-2} d^{-1} for all nutrients.

Statistical analyses

For lysimeter concentration and flux data, we tested differences among vegetation types using a repeated-measures ANOVA, with Type II sums of squares as a conservative approach to handle our unbalanced design and missing data. Our unit of observation was individual trees, rather than species means. We calculated pairwise contrasts among turfgrass,

evergreens, and deciduous trees evaluated with the Holm procedure (overall α=0.05) to control the strong familywise error rate for multiple planned contrasts.

To test for relationships between soil water nutrient concentrations and tree traits, we used quantile regression (Koenker 2001). Quantile regression is able to handle datasets with non-constant variance (often resulting in "triangular" scatterplots in ecological data) and outliers better than ordinary least-squares regression, and also adds the ability to explore relationships at other parts of the distribution besides the mean (Cade2003). We calculated quantile regressions at quantiles (τ) from 0.01 to 0.99 in increments of 0.01. At each τ, we also calculated 90% confidence intervals for slope and intercept estimates using an inverted rankscore test (Koenker 1994) to determine statistical significance.

We present the results of quantile regression graphically as a plot of slope or intercept coefficient across all τ values, plus an envelope of its confidence intervals. Because each quantile regression tests a bandwidth of τ values around the target τ, we followed the convention of plotting the coefficients as stair-step plots instead of lines connecting discrete points. Although many other authors plot fewer points to smooth the confidence intervals (usually τ increments of 0.05), for our smaller datasets this sometimes resulted in obscuring segments where the confidence interval fluctuated back and forth across zero. We chose to err on the side of full disclosure and plotted the confidence intervals at every τ increment of 0.01, like the coefficients, even though this resulted in more jagged plots. Selected statistically significant regression lines were also plotted on the data scatterplot. These graphs allow us to display trends in the coefficients increasing or decreasing systematically at different quantiles of the distribution.

We focused on identifying trait relationships that explained not only a central tendency in the lysimeter data (quantiles around the median) but also in the upper quantiles, i.e. those trees that have the highest nutrient leaching to groundwater. While relationships at any quantile are scientifically interesting and helpful for understanding underlying mechanisms, relationships at upper quantiles explain large differences across the range of the predictor variable in trees with the highest leaching. Relationships at low quantiles, by contrast, explain relatively small differences among trees with low leaching. A trait relationship that predicts which trees are likely to have very high leaching can generate management recommendations

that advise planting fewer of these trees in favor of others that better decrease nutrient leaching.

We performed all statistical analyses in R 3.0.2 (R Core Team, Vienna, Austria), using base packages plus ANOVA functions from ez 4.2-2 and quantile-regression functions from quantreg 5.05.

Scaling up

We scaled up our results to the Capitol Region Watershed (~17,400 ha), which has a land cover of 33% tree canopy cover and 22% low-statured vegetation (Kilberg and Bauer 2011). We treated all low-statured vegetation as open turfgrass, and estimated the proportion of deciduous and evergreen trees using data from a survey spanning public and private lands in Minneapolis (immediately west of the Capitol Region Watershed), which found 89.9% deciduous trees and 10.1% evergreen trees by canopy area (Nowak et al. 2006a). Because some tree canopy overhangs impervious surfaces, we used QuantumGIS 1.8 (QGIS Development Team, Open Source Geospatial Foundation) to clip the tree canopy maps to exclude buildings, streets, and alleyways mapped by the Captiol Region Watershed District (*unpublished data*). Sidewalks and other smaller impervious surfaces were not mapped, so we slightly overestimated the area of tree canopy over pervious surfaces. The land-cover classification by Kilberg and Bauer did not distinguish between deciduous and evergreen trees; we assumed that all tree canopy cover over streets, alleyways, or buildings was deciduous trees because evergreens are only planted as boulevard trees in wide boulevards and parkways where they typically do not overhang the street (Saint Paul Parks and Recreation 2013), and they generally have pyramidal forms rather than high spreading canopies that overhang buildings.

In scaling up our results, we assumed that our lysimeters sampled an area representing average soil water nutrient concentrations under a tree's canopy. We averaged together all deciduous or all evergreen trees in our study; if future work is better able to resolve species differences, it will be possible to weight the averaging by each species' proportion of total canopy area in the watershed (Nowak et al. 2006a).

We estimated the ecosystem service value of trees reducing P leaching to groundwater by

using the cost to remove the same amount of P with stormwater infrastructure. The Capitol Region Watershed District measured the P removed by several different stormwater "best management practices" and generated cost-benefit comparisons. Annual costs per kg P removed were $1,570 for a regional stormwater pond, $4,200 for infiltration trenches, and $6,140 for rain gardens (Capitol Region Watershed District 2010). We used the cost/kg for infiltration trenches to estimate a dollar value for trees reducing P leaching to groundwater because infiltration trenches were the most cost-effective option that, like trees, could theoretically be used throughout the entire watershed. The regional stormwater pond was more cost-effective, but regional-scale ponds could only be installed in a select few locations serving a small portion of the watershed, making it a less-useful comparison.

Results

Soil water nutrient concentrations

Below-normal summer and fall precipitation (Figure 2.1) frequently left the soil too dry to obtain soil water samples during mid- to late summer through fall, with dry soils persisting through the following early spring, of all sampling years. Successful sampling dates spanned 7/28-9/1/2011, 4/26-7/13/2012, and 5/16-8/2/2013. We were not able to obtain water from every lysimeter on every sampling date, especially towards the beginning and end of sampling periods each year. Unfortunately, soils were too dry to obtain soil water samples during leaf-out or litterfall periods in any study year; as a result, we were not able to address our hypothesis about different seasonal patterns between evergreen and deciduous trees.

For both N and P, deciduous trees had lower soil water nutrient concentrations than open turfgrass areas (Table 2.1, Figure 2.2). By contrast, evergreen trees had lower soil water P concentrations than turfgrass, but similar N concentrations. N concentrations showed a seasonal pattern, with high concentrations in spring 2012 and 2013 declining rapidly, but P showed no clear seasonal pattern. There were significant differences among vegetation types: For N, deciduous trees < evergreens ≈ turfgrass; and for P, deciduous trees < evergreens < turfgrass (Table 2.1). There was also noticeable interannual variability, especially between 2012 and 2013.

N and P concentrations in soil water samples were essentially uncorrelated and independent (Figure 2.3). On average, 60% of both total N and P were present in inorganic forms (NO_3^- and SRP). Samples with higher total N or P also had a statistically significant tendency to have a higher percent as inorganic N or P, respectively (Figure 2.4). 88% of soil water samples with N concentration over 2mg/L and 86% of soil water samples with P concentrations over 100µg/L were more than 50% inorganic. The percent of N as NO_3^- showed a highly bimodal distribution; 21% of the lysimeter samples had less than 1% of the N as NO_3^-, and 19% had more than 99% of the N as NO_3^-. In contrast, the percent P as SRP showed a relatively even distribution.

The two sites where *F. pennsylvanica* trees had been removed had high pulses of soil water N in spring 2012, following removal. In spring 2013, their soil water N concentrations were higher than average for deciduous trees but very comparable to a neighboring still-living *F. pennsylvanica* tree. These sites did not show a similar pulse of P after removal, and their soil water P concentrations throughout 2012 and 2013 were similar to their neighboring live conspecific.

Our analyses excluded N data from under one evergreen (*A. concolor*) that had suspiciously high N concentrations in its soil water samples, especially during late summer 2011 (N concentrations 80-110mg/L) when all other soil water samples had low N concentrations (the average for other evergreens was <5mg/L). We cannot rule out contamination of the lysimeter or the soil, and so chose to exclude these data. P concentrations from under this tree were within the range of other samples, so we kept those data in our analyses.

Water and nutrient fluxes

Modeled estimates of daily vertical water fluxes at 60cm showed episodic pulses with dampened responses to rainstorms that tailed off more gradually than precipitation (Figure 2.5). During periods when BROOK90 predicted that the upper soil layers were drier than deeper soil layers to the extent that the upwards matric potential gradient was stronger than gravity, it predicted a small upwards (negative) soil water flux. We included these negative values when calculating average or total nutrient fluxes, as we believe that they represent a real upwards wicking of soil water and nutrients

Overall, BROOK90 predicted lower vertical water fluxes under turfgrass than under most trees, driven by the trees having lower ET than turfgrass, with the exception of two high-LAI evergreens (both *A. concolor*). Because trees had lower soil water nutrient concentrations than turfgrass, while water fluxes showed an opposing pattern of higher water fluxes under trees, our estimated differences among functional types' nutrient fluxes were smaller than the differences among their soil water nutrient concentrations.

Total (2011-2013) N leaching (Table 2.2) was statistically similar between deciduous trees and turfgrass despite statistically discernable differences in soil water N concentrations, though still suggestive of lower leaching under deciduous trees. Total leaching was discernably higher for evergreens, driven by the predicted higher water flux under evergreens than turfgrass. Seasonal patterns of N leaching, and patterns among vegetation types, were also substantially different among years (Table 2.2, Figure 2.6), largely driven by different patterns of soil water N concentration. Total P leaching was discernably different among the three vegetation types, driven by the pattern in soil water P concentrations that was large enough to offset the opposing differences in water fluxes: deciduous trees < evergreens < turfgrass (Table 2.2, Figure 2.6).

Tree trait relationships

We tested relationships between soil water N and P concentrations and various tree physiological traits singly and in combination: leaf, litter, and root chemistry, growth rate, leaf biomass, projected canopy area, and LAI. Although we had also planned to test trait relationships with total annual nutrient fluxes, our estimates of individual trees' fluxes span different lengths of time because individual lysimeters often did not yield water at the beginning or end of each year's sampling period.

We divided our sampling dates into "spring" and "summer" seasons because of the strong seasonality in soil water N concentrations. We classified the first two sampling dates in both 2012 and 2013, with high and sharply declining N concentrations, as "spring" seasons and all others as "summer." To limit the effects of missing data (cases where a lysimeter did not yield water on that sampling date, or yielded too little water for all chemical analyses), we first averaged together samples within a season (i.e. summer 2011, spring 2012,

summer 2012, spring 2013, summer 2013) because concentrations were more similar within a season than among seasons. We then averaged together the two spring season means to create an overall mean spring value, the three summer means to create an overall mean summer value, and all five seasonal means to produce a grand mean. We tested for relationships with tree traits using both seasonal means as well as the grand mean for response variables, testing evergreen and deciduous trees separately.

For N, we found different relationships during spring and summer. Springtime soil water N concentrations among deciduous trees showed a negative relationship with root %N (Figure 2.7). It should be noted, however, that the five trees with high root %N (>1.5%) that drove this pattern were all *C. occidentalis*. Among evergreens, springtime soil water N concentrations showed positive correlations with any of a well-correlated set of predictors: leaf biomass, canopy N mass (green leaf %N x iTree-estimated leaf biomass), or litter N mass (litter %N x leaf biomass). Variation among trees' leaf or litter %N was much smaller than variation among their leaf biomass, so variation in leaf biomass accounted for most of the variation in all three predictors (Figure 2.7). Summertime soil water N concentrations did not show statistically significant relationships with tree traits, though they are suggestive of similar relationships as for springtime concentrations. We excluded the *G. triacanthos* N-fixer, which was an outlier with higher soil water N concentrations than any other tree in our study, from regressions with soil water N concentrations.

For P, we found similar relationships in both spring and summer, so we present trait relationships using the grand mean. Both deciduous and evergreen trees showed a statistically significant relationship with litter C:P ratio, though in opposite directions: positive for deciduous trees, and negative for evergreen trees (Figure 2.8). In both cases, the relationship was noticeably stronger (steeper slope) at higher quantiles. Litter P concentration and litter C:P ratio were tightly correlated for both evergreen and deciduous trees, and quantile regressions using either predictor gave very similar results. We chose to present litter C:P because it does a noticeably better job of explaining the two deciduous trees with highest soil water P concentrations

Neither xylem anatomy (gymnosperm, ring-porous, diffuse-porous), mycorrhizal type (arbuscular mycorrhizae, ectomycorrhizae), nor LAI appeared to explain any of the residual

variability in our trait relationships for N or P concentrations.

Soil nutrients

Average soil bulk %C and %N declined with depth, from 3.8 %C and 0.30 %N in 0-10 cm to 1.1 %C and 0.06 %N in 40-60cm. In the 0-10 cm samples for extractable nutrients, KCl-extractable inorganic N was 0.032 mg N/g soil, the average 10 d net mineralization rate was 0.024 mg N/g soil, and the average Brays-P was 0.0094 mg P/g soil. We found no significant differences among vegetation types for soil nutrient pools or net mineralization or nitrification rates. (We had also originally planned to repeat Brays-P, extractable-N, and net-mineralization and -nitrification measurements during each spring, summer, and fall of our sampling period, but the drought prevented this as well.)

There was a positive relationship at central and upper quantiles (τ = 0.36 to 0.96) between net mineralization rate and springtime soil water N concentration, and a weaker positive relationship with summertime soil water N concentration (again excluding the removed trees and the N-fixer). Net nitrification showed a weaker positive relationship with springtime soil water N concentration at some central and upper quantiles, and no significant relationship with summertime soil water N concentration. For P, there was a positive relationship across nearly all quantiles (τ = 0.04 to 0.96) between Brays-P and soil water P concentrations. Including tree traits and soil nutrients together in quantile regressions did not improve their predictive power.

Scaling up

Of the 5,740 ha of tree canopy cover in the Capitol Region Watershed (Kilberg and Bauer 2011), we estimated that 4,870 ha (74.8%) is deciduous canopy over pervious surfaces, and 579 ha (10.1%) is evergreen canopy over pervious surfaces. Multiplying by our average fluxes for each vegetation type, we estimated that trees reduced P leaching to groundwater by 533 kg ($2.24 million worth) in 2012 and 1201 kg ($5.04 million worth) in 2013. For homeowners or other decision-makers thinking on the scale of individual trees rather than thousands of hectares, a typical mature deciduous tree in our study had approximately 100 m^2 of projected canopy area, and a typical mature evergreen had approximately 50 m^2. We

estimated that a deciduous tree of this size reduced P leaching to groundwater by 1.1 g ($5 worth) in 2012 and 2.3 g ($10 worth) in 2013 relative to turfgrass, and an evergreen of this size reduced P leaching to groundwater by 0.1 g ($0.40 worth) in 2012 and 0.7 g ($3 worth) in 2013.

Discussion

Nutrient leaching to groundwater

N and P concentrations in soil water were high enough to degrade water quality: 40% of lysimeter samples exceeded 50 μg P/L, which is the average of deep- and shallow-lake eutrophication standards for the ecoregion (MN Administrative Rules, Ch. 7050) and 17% exceeded the 10 mg NO_3-N / L standard for drinking water (MN Administrative Rules, Ch. 4717). Although there are not formal eutrophication standards for total N in Minnesota surface waters, our soil water samples often had higher total N concentrations than would be typically found in agricultural streams (national median = 4 mg/L, Dubrovsky2010).

Nutrient leaching to groundwater also has the potential to be a substantial contributor to nutrient loading in the Capitol Region Watershed. To generate a first-order estimate, we multiplied our leaching fluxes by the total watershed area of turfgrass and deciduous or evergreen tree canopy over pervious surfaces. We estimated that nutrient leaching contributed 507 kg/km^2 N and 6.3 kg/km^2 P in 2012, and 990 kg/km^2 N and 15.0 kg/km^2 P in 2013. For comparison to inputs, rates of atmospheric deposition for our study area are estimated to be 1080 kg N and 24.7 kg P km^{-2} yr^{-1} (Fissore2011 and Barr1994); we estimate that leaching was equivalent to 47% of atmospheric N inputs and 26% of P inputs in 2012, and 92% of N inputs and 61% of P inputs in 2013. These comparisons overestimate the proportion of inputs that leach to groundwater, however, since they do not include inputs from weathering, N fixation, or pet waste. In addition, the estimates of N deposition are derived from "background" deposition measurements at a rural site 50km north of our study sites, and likely underestimate urban N deposition.

For comparison to stormflow, Janke and coauthors (2013) found stormflow in different subwatersheds of the Capitol Region Watershed contributed 160-500 kg/km^2 total N

(dissolved + particulate) and 20-80 kg/km^2 total P (dissolved + particulate) during the warm season (May 1 – Oct. 31; averages of data from 2006-2011). Our data suggest that leaching to groundwater from pervious surfaces could contribute more N loading than stormwater in the watershed, and over 10% as much annual P loading as stormwater. Janke and coauthors also found that baseflow in storm drains, which comes largely from shallow groundwater leaking into the drains, contributes a similar amount of N and P as our estimates of leaching.

Shallow groundwater in the Capitol Region Watershed shows elevated concentrations of N and P. Samples from 2012 of groundwater-dominated springs, drains, and storm-drain baseflow had average concentrations of 3.03 mg N/L and 22.13 μg P/L (J. C. Finlay and B. Janke, *unpublished data)*. These concentrations were on the low end of our lysimeter soil water samples, likely due to not only removal/retention below 60 cm but also dilution and mixing with deeper groundwater that has lower nutrient concentrations. The average N:P ratio of these groundwater samples was 178, compared with 310 for our lysimeter soil water samples, suggesting 1) higher rates of denitrification and/or N retention than P retention in subsoils and groundwater, and/or 2) additional P inputs below 60 cm from weathering of P-rich sandy subsoils.

Our findings that a substantial percentage of soil water P is in inorganic forms (SRP) was not expected. Because SRP is fairly immobile in many soils, we had speculated that high concentrations of P in soil water might exist largely as organic P, in molecules that do not adsorb strongly onto soil mineral surfaces. While there may have been some mineralization of organic P in the lysimeters, our findings clearly do not support a conceptual model where P leaches primarily in organic forms.

Uncertainties in hydrologic modeling

Our predicted differences in water fluxes under urban trees and turfgrass, where trees have lower ET and higher leaching, have not (to our knowledge) been confirmed by empirical data. Although we calibrated the BROOK90 hydrologic model to local, empirical data for urban tree transpiration and turfgrass ET, we were unable to find empirical data for ET of a combined system with a tree canopy over a turfgrass understory. The BROOK90 model has been used in non-urban savannas (e.g. Dijkstra et al. 2006), but it only models a

single canopy layer and is not explicitly designed to model vegetated understories. As such, its outputs for a combined tree-turfgrass system are more uncertain. Peters and coauthors (Peters et al. 2011) modeled an urban combined tree-turfgrass system and estimated that its ET is lower than open turfgrass ET, but their model assumed that the turfgrass under a tree canopy is shaded throughout the day. In reality, however, for an urban tree without limbs below ~2m, its shadow falls under its canopy for only part of the day, when the sun is more overhead; at other times, its shadow falls on nearby areas and the turfgrass under the canopy receives direct sunlight. Although we parameterized the BROOK90 model in an attempt to simulate a partially-shaded turfgrass understory, without empirical data for verification these outputs contain considerable uncertainty.

If urban trees with turfgrass understories instead have higher ET and lower water fluxes than open turfgrass areas, it would lower our estimates for nutrient leaching under trees and change the patterns among vegetation types. We would still find the same pattern for P leaching, where turfgrass > evergreens > deciduous trees, though the difference between trees and turfgrass would be larger than our estimates. For N, we expect that we would find statistically significant lower leaching under deciduous trees, amplifying the pattern in soil water N concentrations. As such, we consider our findings for P leaching robust against the uncertainties in our hydrologic model, while our findings for N leaching are much more sensitive.

Uncertainties in scaling up

Our watershed-scale estimates of both total leaching and tree effects contain considerable uncertainty. We assumed that tree effects are proportional to the projected canopy area over pervious surfaces; however, tree rooting zones generally extend well beyond the canopy edge (Crow2005), so tree influences may affect a larger area than in our calculations. Furthermore, we did not investigate the spatial heterogeneity in soil water N and P concentrations under a tree to understand either random heterogeneity or systematic variation with distance from the trunk, so we have no data to test our assumption that concentrations measured halfway between the trunk and canopy edge represent an average value. In scaling up to the watershed, we further assumed that all ages and sizes of trees have similar leaching rates per

m^2 of canopy area, although we only studied mature trees. These factors highlight the uncertainty associated with using our data on open-grown trees in parks to estimate nutrient leaching under trees in other situations such as boulevard trees, which comprise a substantial portion of the urban forest. Trees near streets or building foundations have constrained and distorted rooting zones compared to open-grown trees, likely leading to different ratios of tree root mass (and hence uptake, root litter inputs, and other root-based processes that can affect nutrient leaching) per unit of canopy area over pervious surfaces. Similarly, any litter- or wood-based processes reducing leaching (litter export, nutrient storage in wood) would not be decreased by having impervious surface under the canopy and may be underestimated by our method excluding tree canopy over impervious surfaces.

Management practices in our study parks are also not representative of the entire watershed. Leaf litter in the parks is mown into the grass and retained on-site, while considerable amounts of leaf litter in other sites are removed from the property, exporting N and P and likely decreasing leaching. Leaf litter that enters the street gutter washes down storm drains or is removed by street sweeping, and only 34% of watershed residents report keeping leaf litter on-site (either composting on their property or mulching into the lawn, Wein2010). We also likely underestimated N leaching in fertilized areas compared to our unfertilized parks; 52% of residential lawns in the watershed are fertilized (Wein2010), as are most city parks in suburbs outside of Saint Paul (E. Peters, *personal communication*). Although we do not have data on commercial, industrial, or institutional landscape management, from our personal observations and conversations with land managers we expect that fertilization is more common than on residential lawns.

Finally, our watershed-scale estimates assume that 100% of nutrient fluxes at 60 cm reach groundwater. While it is likely that biotic and abiotic interactions in the soil would reduce N and P concentrations between 60 cm and the water table, our study sites' combination of coarse sand subsoils, P-rich mineralogy, and shallow water table lead us to expect that the bulk of the soil water N and P at 60 cm will reach groundwater and substantially contribute to nutrient loading in local lakes and the Mississippi River.

Tree effects

We found that trees in grassy areas reduced P leaching to groundwater, with lower leaching under deciduous than evergreen trees. In contrast, our data do not give a clear answer whether trees increase or decrease N leaching, since trees had lower leaching than turfgrass in 2012 but higher leaching than turfgrass in 2013. Much of the N leaching took place during high springtime pulses, the beginnings of which we likely did not sample completely because both the 2012 and 2013 growing seasons began with a period of rewetting after severe drought that made it difficult to obtain lysimeter samples. In addition, the magnitude of these pulses and the differences among vegetation types may also be different during normal hydrologic years than during the post-drought rewetting we observed, as drying/rewetting cycles have been shown to increase N leaching (Gordon et al. 2008). We recommend sampling at least one spring in non-drought conditions to better understand tree effects on N leaching in more normal hydrologic conditions.

Trees can reduce P leaching either by reducing P inputs and/or increasing P storage; we consider it more likely that trees increased P storage. P inputs to our unfertilized study sites were primarily from atmospheric deposition, soil mineral weathering, and pet waste. Although tree leaves intercept P-containing fine dust, this would not have reduced inputs since the P either washed off leaves during rain events or became part of leaf-fall. We consider it unlikely that trees substantially reduced mineral weathering rates relative to turfgrass, since soil water fluxes were slightly higher under trees, and because the increased root mass and rooting depth of trees is likely to have led to more root exudates and a slight increase in weathering. We also consider it unlikely that pet waste inputs were substantially lower under trees than in open grassy areas, given the propensity of dogs to urinate on vertical objects such as tree trunks. (This is in contrast with N inputs: we observed less clover growing in the shade of trees, suggesting that trees could reduce N inputs by decreasing N fixation.)

Trees can increase P storage in soil or wood. Although we did not directly measure soil P, due to the difficulty in separating biologically-meaningful fractions, we examined soil nutrient storage through relationships with litter chemistry. If trees were increasing soil storage of P by accumulating P in soil organic matter, we hypothesized that trees with lower

litter nutrient concentrations would have slower-decomposing litter, higher storage of nutrients in soil organic matter, and lower nutrient leaching. Although we found significant relationships between soil water P concentrations and litter C:P ratios, there was a positive relationship among deciduous trees and a negative relationship among evergreens. Because the relationship among deciduous trees is opposite to our hypothesis, our data do not offer clear support for trees increasing soil storage of P.

To estimate P storage in tree wood, we used i-Tree estimates of annual tree growth, reported in kg C/yr as "gross C sequestration." We assumed a constant C:P ratio of 1533.3 (Fissore et al. 2011), since species-specific estimates are not available, to estimate that average annual P storage in tree wood was 9.9 g for deciduous trees, and 3.5 g for evergreens. This is substantially larger than our estimates of the amount by which an individual typical-sized tree reduced annual P leaching fluxes relative to turfgrass (deciduous: 1.1 g in 2012, 2.3 g in 2013; evergreen: 0.1 g in 2012, 0.7 g in 2013), suggesting that P storage in wood is large enough to account for the observed reductions in P leaching. However, we did not find a statistically discernable relationship between growth rate and soil water P concentrations (data not shown), so there is no evidence that differences in P storage drive differences among trees within vegetation types.

Alternative mechanisms

Differences in P concentrations and fluxes between trees and turfgrass do not necessarily demonstrate that trees reduced the amount of P leaching to groundwater. We also examined three alternative mechanisms that could also produce differences between trees and turfgrass, but in which trees do not actually reduce P leaching to groundwater: 1) Trees establish preferentially in microsites with different abiotic conditions; 2) Trees spatially redistribute nutrients to turfgrass areas, creating spatial heterogeneity but no net reduction in leaching; and 3) Trees store nutrients during the growing season, temporarily reducing leaching, but release those nutrients to be leached during the fall.

The first alternative reverses cause and effect, where pre-existing abiotic differences favor tree establishment in microsites that also have lower P leaching. Trees in savannas and grasslands have been shown to establish preferentially in microsites with favorable soil

properties (e.g. texture, pH, base saturation; Geiger et al. 1994). Differences in nutrient cycling beneath trees in these systems are influenced by pre-existing abiotic differences as well as the effects of the trees themselves. Furthermore, plants may generate positive-feedback loops that can reinforce pre-existing high or low levels of nutrient availability (Hobbie 1992).

In urban systems, however, most of the trees were planted by people, and their specific planting locations are selected for reasons (e.g. aesthetics, salt tolerance, overhead wires) that have little to do with microsite patterns of nutrient cycling. While it is still rare to find controlled, randomized experiments incorporated into landscape design (but see Nassauer and Opdam 2008), urban systems provide an environment where biotic and abiotic drivers are easier to disentangle. We do not believe that pre-existing abiotic differences are responsible for the differences we observed between trees and turfgrass.

The second alternate mechanism is that the observed differences are due to spatial redistribution of P. Deciduous leaf litter blows around extensively in open parks and lawns, and in doing so drives a net export of soil nutrients from a tree's rooting zone (since there is not an equivalent import of grass clippings from open turfgrass areas). Litter export would be expected to decrease nutrient leaching beneath trees, but at the expense of increasing nutrient leaching in turfgrass areas that receive the litter. The same amount of P could be leaching to groundwater across the watershed, but redistributed from trees to turfgrass.

While we observed litter redistribution, our tree trait relationships suggest that this was not a primary driver of the differences we found in soil water P concentrations. If litter redistribution were a primary driver, we would expect to have seen negative relationships between soil water nutrient concentrations and the total mass of P in deciduous leaf litter (leaf biomass x litter P concentrations). To separate this conceptual model from one in which litter chemistry alone has a negative relationship, through a mechanism such as affecting decomposition rates, if litter redistribution were a primary driver we would expect leaf biomass to also have a significant negative relationship with leaching. Among trees with similar litter P concentrations, those with a greater mass of litter would export more P via litter redistribution and hence have lower soil water nutrient concentrations. While deciduous trees did show a negative relationship between soil water P concentrations and the total mass

of P in litter, this was driven by leaf chemistry and not also by leaf biomass. In a regression with litter P concentration and leaf biomass (data not shown), leaf biomass was only statistically significant at quantiles $\tau < 0.24$, explaining relatively small differences among trees with low soil water P concentrations. At central and upper quantiles, which are of greater interest, leaf biomass was not a significant predictor of soil water P concentrations. Because leaf biomass does not explain differences among trees with similar litter P concentrations, our data do not support an alternate mechanism of litter redistribution driving the differences we observed between trees and turfgrass.

The third alternate mechanism is a temporal redistribution, where lower nutrient leaching during the growing season is driven by a temporary storage of P in the trees' canopy. Reduced leaching during the spring and summer could be offset by increased leaching after litterfall, when some of the P stored in the canopy are returned to the soil. Because the drought limited our sampling to spring and early summer, our data do not directly rule out this possibility.

While we expect to see seasonal patterns in non-drought years, with higher nutrient leaching after litterfall than during spring and summer, we do not expect that this would offset overall leaching differences between trees and turfgrass. First, because deciduous leaf litter blows around in these systems, nutrients released by decomposing litter are not highly concentrated beneath deciduous trees. Any increase in autumn leaching (compared to spring/summer leaching) from nutrients released from litter would likely be similar under deciduous trees and in open turfgrass areas (by contrast, evergreen needle litter tends to stay beneath the tree and could lead to a localized increase). Second, if the differences we observed in spring/summer leaching between trees and turfgrass were driven by trees temporarily storing nutrients in their canopy, we would have expected to see negative relationships between soil water P concentrations and the total mass of P stored transiently in the canopy and dropped in leaf litter. In this conceptual model, we would again expect leaf biomass to be a significant predictor: among trees with similar litter P concentrations, those with a greater mass of litter would take up and store more P transiently in the canopy and have lower spring/summer soil water nutrient concentrations. As discussed above, however, for deciduous trees there is not a negative relationship between leaf biomass and

soil water P concentrations. Evergreen trees show no statistically discernable relationship between soil water P concentrations and the mass of P in litter. Third, the fact that soil water P concentrations remained low after tree removal suggests that lower P leaching under trees is persistent, at least for a couple of years, even when there is no longer a tree actively taking up nutrients. Further study during non-drought years will be able to directly address questions of temporal redistribution by measuring nutrient leaching across the entire growing season.

Overall, our data do not support any of these three alternative mechanisms that could produce the differences we observed among vegetation types. While spatial or temporal redistributions may play some role in creating the observed patterns between trees and turfgrass, they do not appear to be the primary drivers. As a result, we believe that trees are causing differences in P leaching to groundwater.

Management implications

Urban trees provided significant benefits in reducing P leaching to groundwater during our study, worth approximately $2-4 million annually across the Capitol Region Watershed. Because soil water P concentrations did not show strong seasonal patterns, we expect that trees will also reduce P leaching in non-drought years. The magnitude of this reduction, however, will likely vary in different hydrologic conditions and may be larger in non-drought years when P leaching occurs over the full growing season.

At this time, we cannot make confident recommendations about which tree species would most reduce nutrient leaching. We found higher than expected within-species variability in both soil water nutrient concentrations and tissue chemistry (Figures 2.7 and 2.8. For the species where we sampled multiple individuals, variation within the species was often larger than the variation among species. We recommend broader sampling of tree tissue chemistry in urban environments to understand the variability within species, and also to resolve differences among species, before making any recommendations favoring certain tree species.

With over 130 tree species identified in the Minneapolis-Saint Paul metropolitan area (C. Fissore, *unpublished data*) and a variety of different management practices that affect nutrient

cycling (in particular fertilization and irrigation), it is important to find a way of readily assessing many more trees than would be practical using lysimeter studies. Trait relationships offer the possibility of leveraging extensive existing plant-trait databases, though care must be taken when basing comparisons on data from a small number of non-urban individuals. The correlations between net N mineralization or Brays-P and soil water nutrient concentrations offer another possible simple means of estimating nutrient leaching under many different species of urban trees. We recommend that further studies of nutrient leaching continue to test correlations with plant traits and soil assays in order to develop protocols that can readily assess many combinations of tree species and management practices relevant to urban-forest decision-making.

Table 2.1: Summary statistics for soil water nutrient concentrations (all samples, all years) by vegetation type.

	Veg. type	Mean	Median	Std. Error	n	significance
Total dissolved N (mg/L)						$p=2.8x10^{-5}$
	turfgrass	7.32	2.56	1.08	94	a
	evergreen	7.07	2.39	0.95	94	a
	deciduous	3.75	1.18	0.55	195	b
	removed	12.38	3.60	3.59	25	
NO_3^--N (mg/L)						$p=8.3x10^{-4}$
	turfgrass	5.63	1.97	1.00	77	a
	evergreen	5.95	1.37	0.97	77	a
	deciduous	2.46	0.25	0.42	146	b
	removed	6.33	1.87	2.40	19	
Total dissolved P (ug/L)						$p=4.5x10^{-12}$
	turfgrass	159.26	72.70	19.88	94	a
	evergreen	84.61	36.94	12.95	97	b
	deciduous	49.86	31.07	4.23	191	c
	removed	24.44	21.03	2.20	25	
Soluble reactive P (ug/L)						$p=6.6x10^{-12}$
	turfgrass	131.23	54.45	19.92		a
	evergreen	59.42	13.55	11.16		b
	deciduous	24.98	12.80	3.12		c
	removed	8.37	7.42	1.61		

Table 2.2: Total nutrient fluxes for each year's sampling period, averaged by vegetation type.

		C		N		P	
2011		(mg/m2)					
	turfgrass	2582.26		273.19		11.536	
	evergreen	11978.13		506.71		5.143	
	deciduous	18744.42		175.82		4.129	
	removed	41551.25		733.93		2.749	
2012							
	turfgrass	2704.95		1496.47		17.531	
	evergreen	15137.85		1488.83		15.209	
	deciduous	11999.68		458.56		6.858	
	removed	8487.53		5160.22		3.409	
2013							
	turfgrass	7136.01		1365.73		40.921	
	evergreen	11796.64		2630.56		26.729	
	deciduous	10897.69		2149.88		17.951	
	removed	13298.12		2366.48		4.000	
Overall			$p=4.44x10^{-8}$		$p=1.15x10^{-8}$		$p=2.14x10^{-13}$
	turfgrass	12423.22	a	3135.38	a	69.99	a
	evergreen	38912.63	b	4626.10	b	47.08	b
	deciduous	41641.78	b	2784.27	a	28.94	c
	removed	63336.91		8260.63		10.16	

Table 2.3: Species codes for Figures 2.7 and 2.8

Code	Species
Ac	*Abies concolor*
Ap	*Acer platanoides*
As	*Acer saccharinum*
Co	*Celtis occidentalis*
Fp	*Fraxinus pennsylvanica*
Gb	*Ginkgo biloba*
Gt	*Gleditsia triacanthos*
Pp	*Pinus ponderosa*
Pr	*Pinus resinosa*
Ps	*Pinus strobus*
Py	*Pinus sylvestris*
Qm	*Quercus macrocarpa*
Tc	*Tilia cordata*

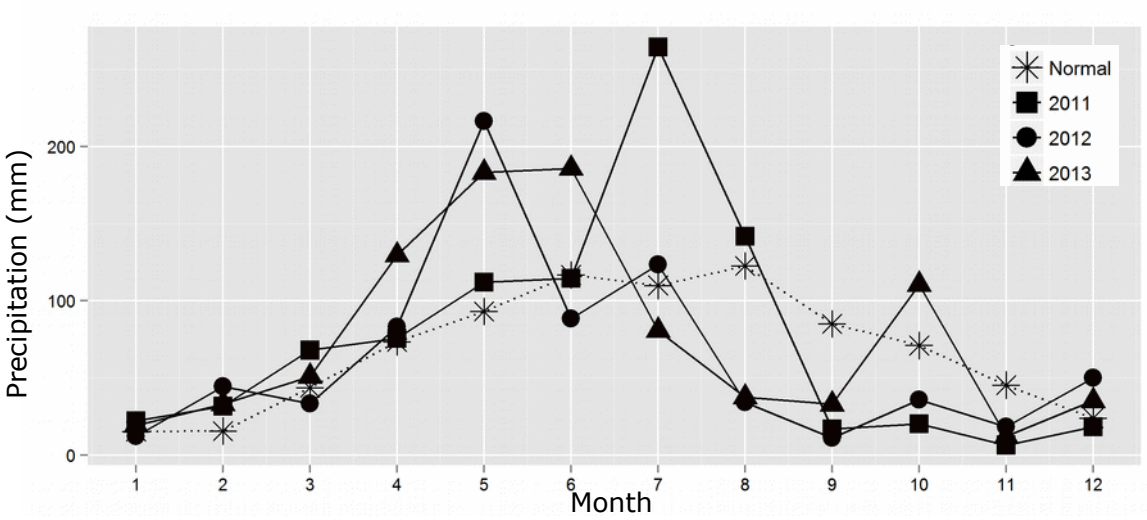


Figure 2.1: Monthly precipitation for 2011-2013 (solid lines) and climate normals (1981-2010, dotted line) for the University of Minnesota weather station (Saint Paul, MN, USA).

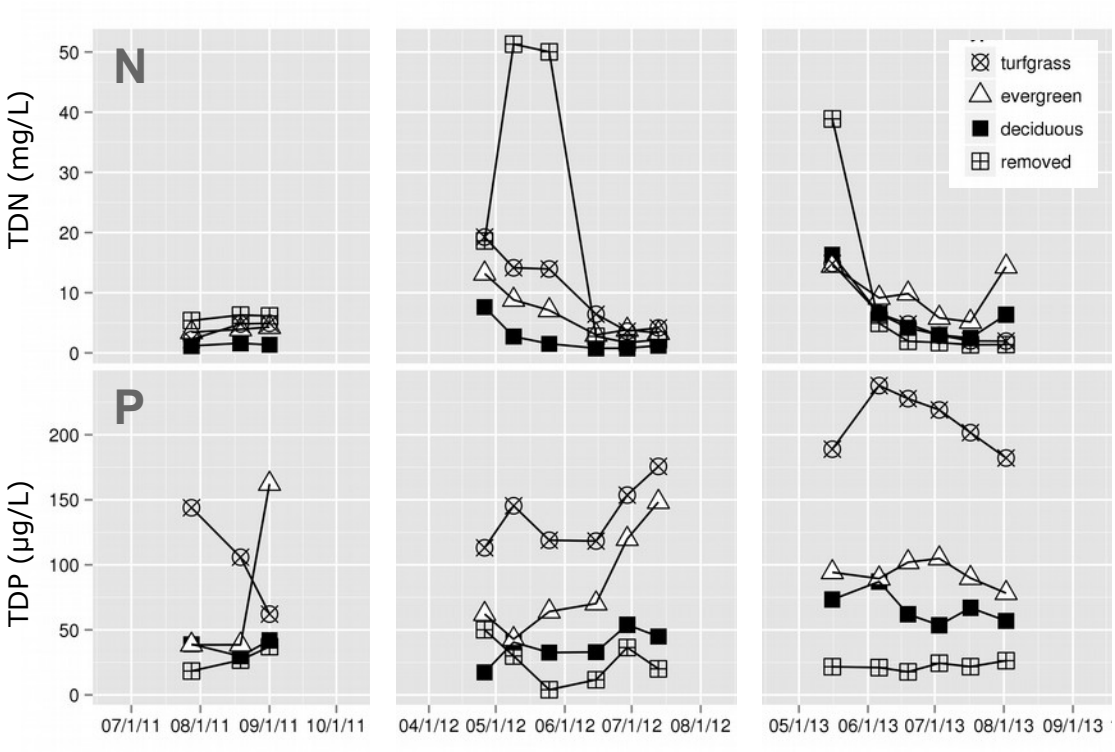


Figure 2.2: Timeseries of soil water total dissolved N (TDN) and total dissolved P (TDP) concentrations, averaged by vegetation type

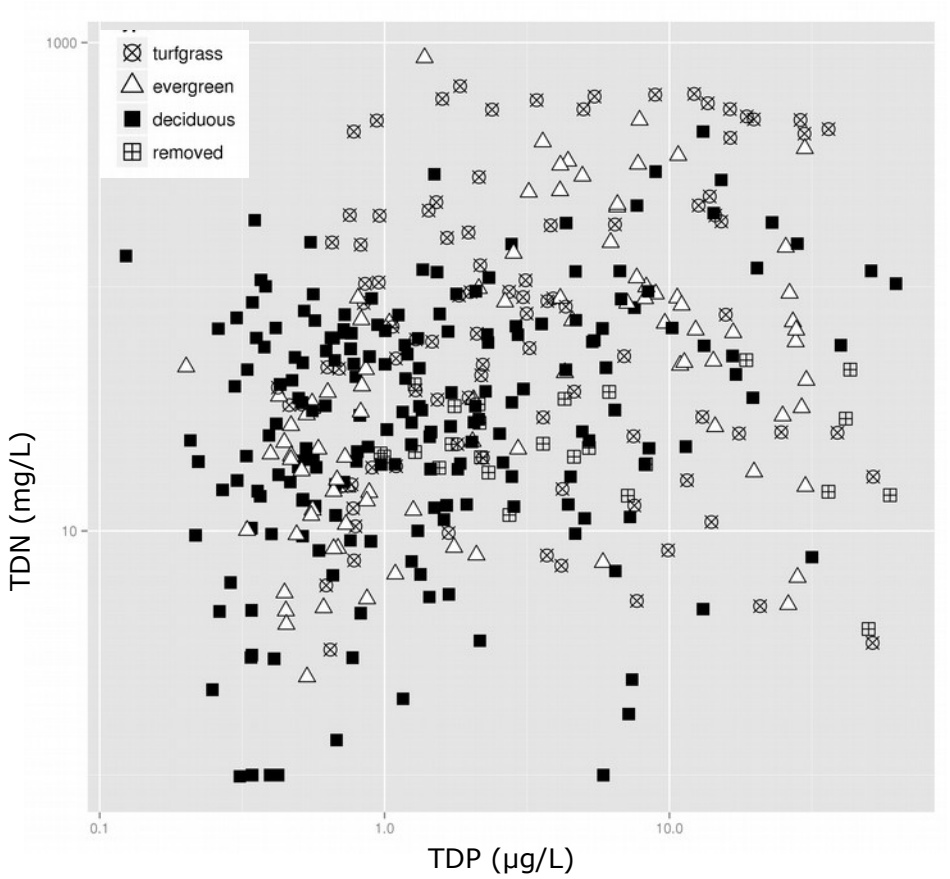


Figure 2.3: Scatterplot of soil water total dissolved N (TDN) vs. total dissolved P (TDP) concentrations

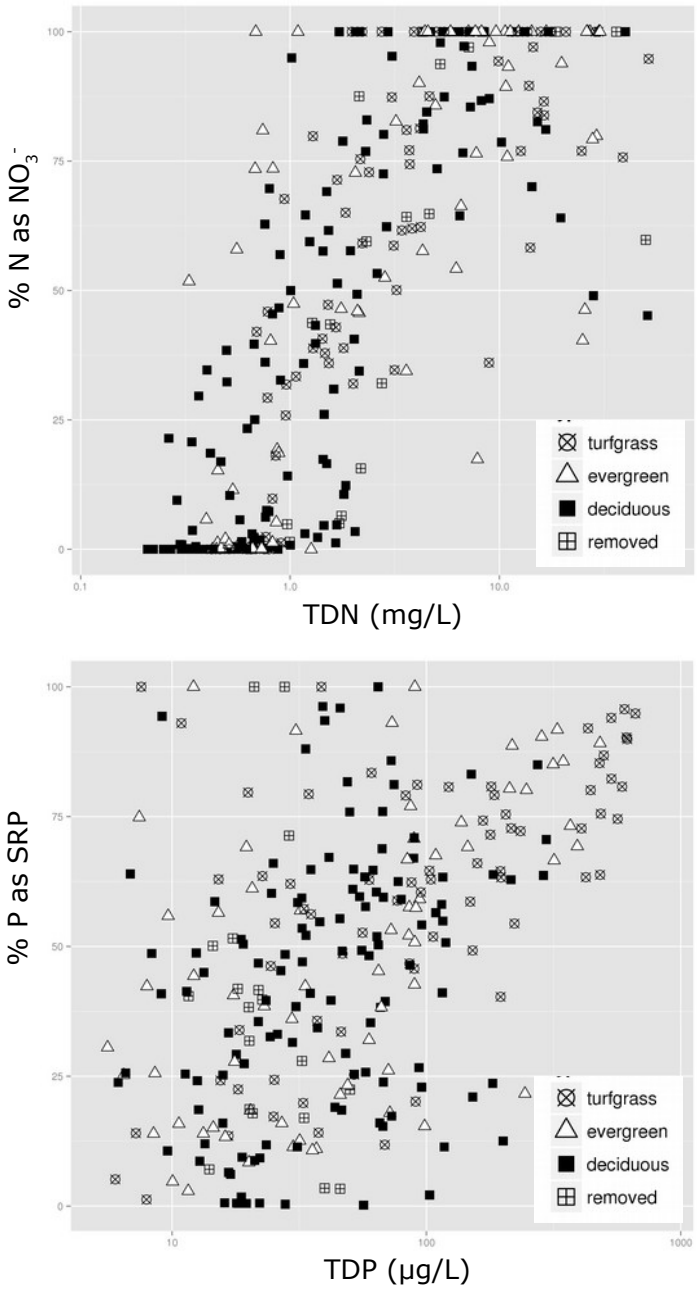


Figure 2.4: Scatterplots of the percent of soil water N and P in inorganic forms (NO_3^- or soluble reactive P (SRP) respectively) vs. total dissolved N (TDN) or total dissolved P (TDP) concentrations. Note log scales on x-axes.

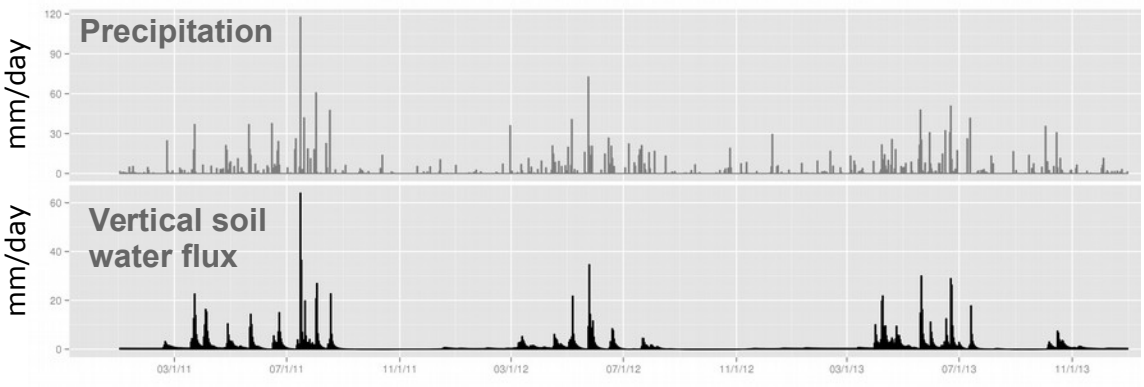


Figure 2.5: Daily precipitation (measured) and vertical soil water flux at 60 cm (modeled) for an *Acer platanoides*

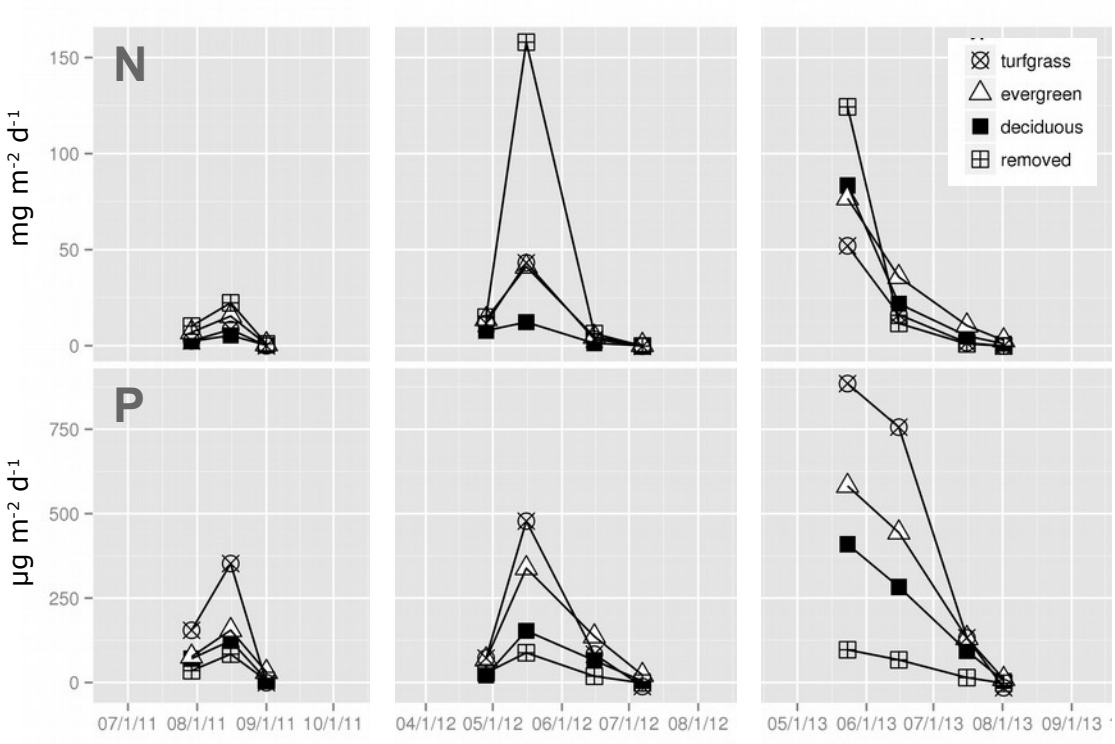


Figure 2.6: Average daily N and P fluxes for each calendar month, averaged by vegetation type

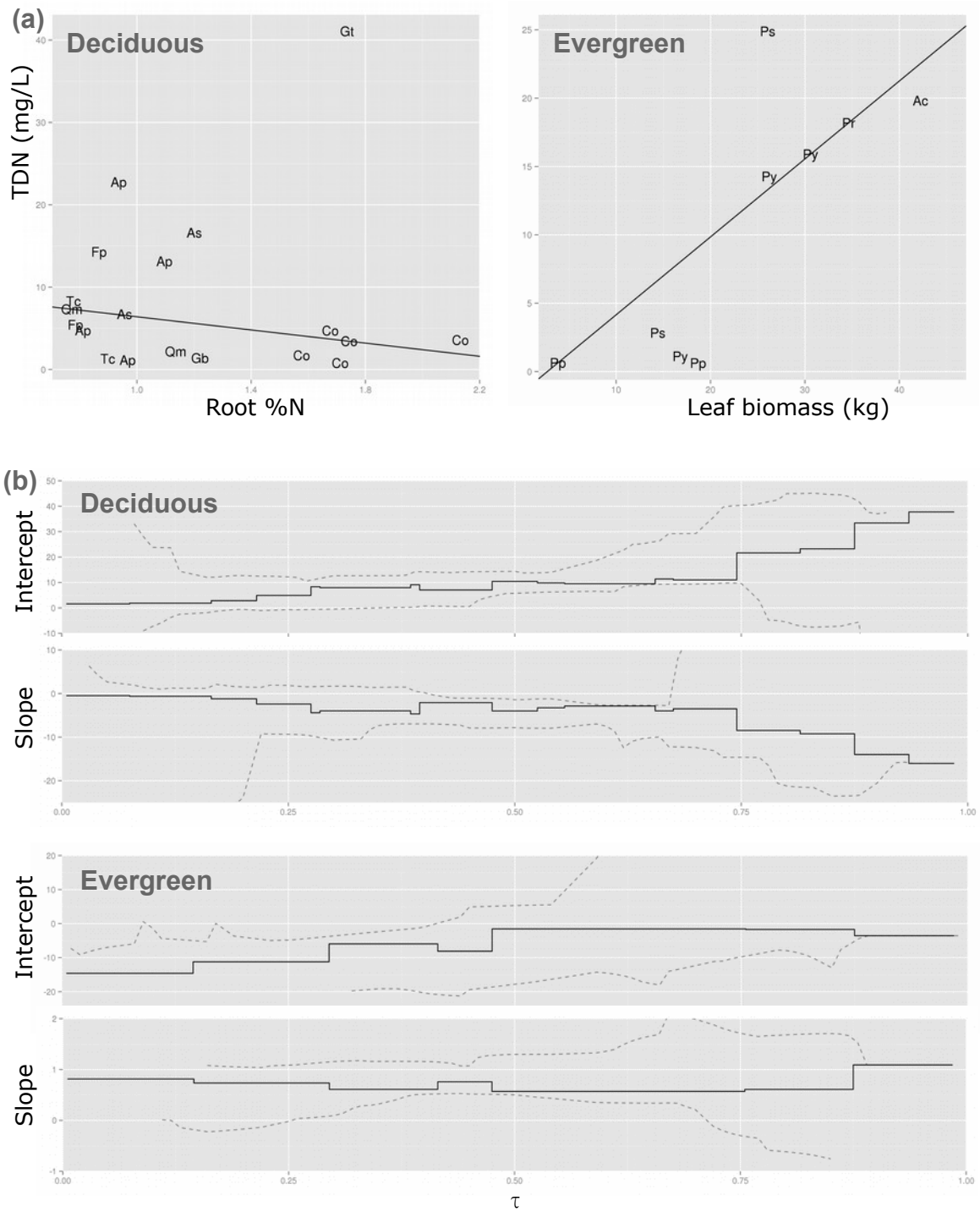


Figure 2.7: (a) Average springtime (first two sampling dates for 2012 and 2013) soil water total dissolved N (TDN) concentrations for each focal tree are best predicted by root %N for deciduous trees, and leaf biomass for evergreens. Solid lines are median regression lines (excluding *Gleditsia triacanthos*). Species codes are listed in Table 3.
(b) Quantile regression graphs, showing intercept and slope across quantiles τ=0.01 to τ=0.99 by intervals of τ=0.01. Dashed lines depict 90% confidence interval envelopes.

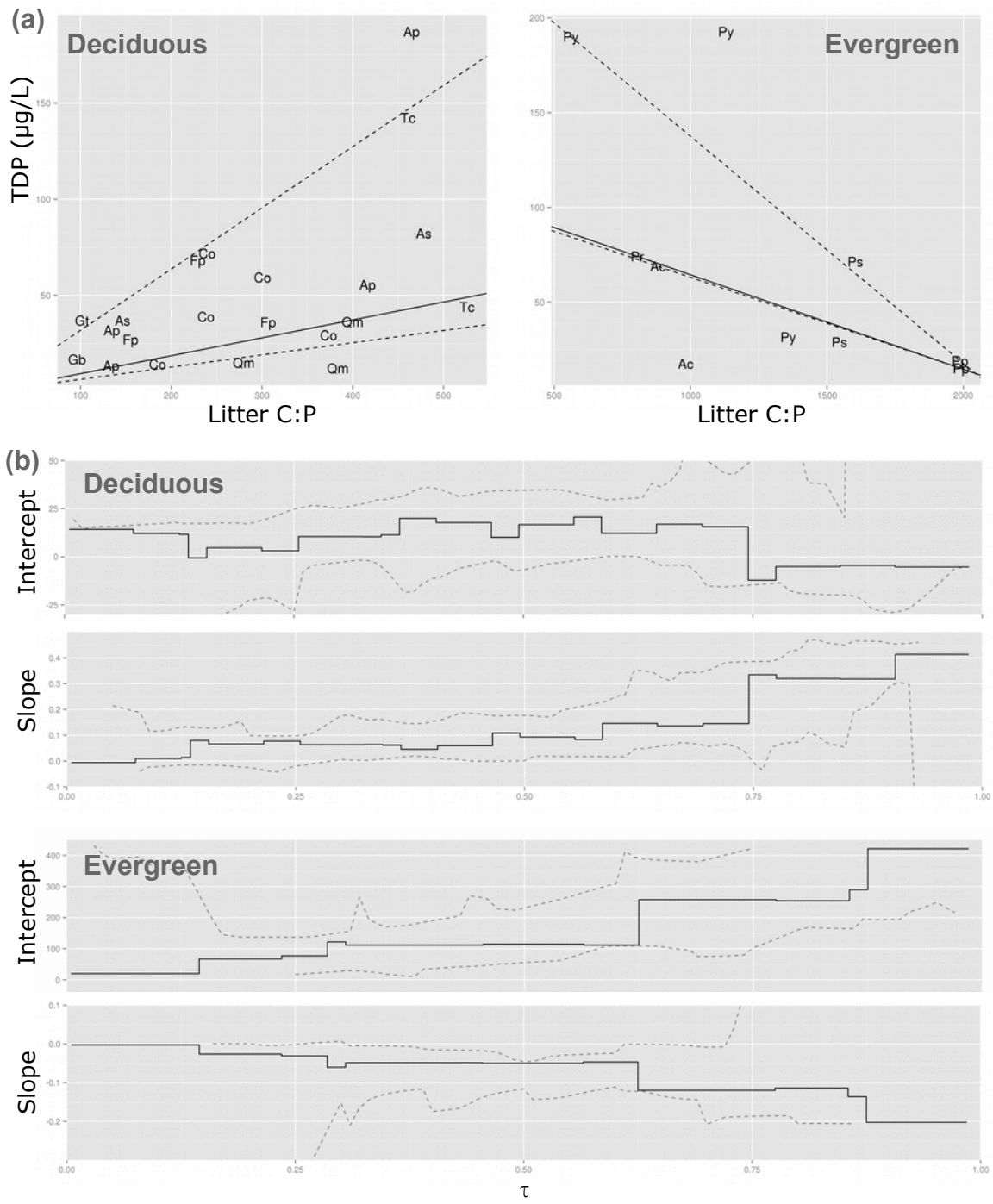


Figure 2.8: (a) Average soil water total dissolved P (TDP) concentrations for each focal tree are best predicted by litter C:P ratios. Solid lines are median regression lines; dotted lines are regression lines near the upper and lower extent of statistical significance. For deciduous trees, these are at quantiles τ=0.3 and τ=0.9; for evergreens, these are at τ=0.35 and τ=0.7. Species codes are listed in Table 3.
(b) Quantile regression graphs, showing intercept and slope across quantiles τ=0.01 to τ=0.99 by intervals of τ=0.01. Dashed lines depict 90% confidence interval envelopes.

Chapter 3:

How do the litter chemistry and phenology of urban trees affect nutrient loading to stormwater?

Abstract

Stormwater flowing over streets, similar to headwater streams, acquires substantial inputs of nutrients from tree litter and other materials. However, many urban streams and downstream water bodies suffer from excess inputs of nitrogen (N) and phosphorus (P), which lead to noxious algal blooms that cause lower water clarity and dissolved oxygen levels, bad odor and taste, and the loss of desirable species. In this study, we examined the role of urban boulevard trees driving N and P inputs to urban street gutters and stormwater. During the 2012 growing season (April-November), we hand-swept biweekly samples of material from the street gutters on city blocks dominated by one of four common tree species that differ in both litter chemistry and phenology (*Acer platanoides, Fraxinus pennsylvanica, Quercus palustris, Tilia cordata*), as well as blocks without tree canopy cover. We size-fractionated these samples (2mm), measured total carbon (C), N, and P, and also leached a subsample in lab as an index of soluble C, N, and P available for transport during rain events.

Differences among tree species in the total amount of nutrients in the street gutters were driven primarily by interspecific differences in the mass of litter dropped, which were much greater than differences in litter chemistry. Autumn litterfall transported 219.0-274.4 kg N km^{-2} and 14.2-20.6 kg P km^{-2} to the street gutters (range is across our study species). Total N and P concentrations decreased during the growing season, driven by changes in the chemistry of litter inputs, while soluble nutrients showed no clear seasonal patterns or differences among species. C and N were less soluble than P; generally less than 5% of total C or N leached out during our lab leaching measurements, in contrast with 10-20% or more of P. N leached out primarily in organic forms, whereas P leached out mainly in inorganic forms. We found that tree phenology is a much more important consideration than litter

chemistry for choosing boulevard tree species and developing management recommendations. Cleaning up spring and autumn pulses of tree litter shortly after they fall has substantial potential to reduce nutrient inputs to stormwater. Because of the wide variation in species' litterfall timing, achieving this goal is likely to require adjusting both boulevard tree selection and litter cleanup strategies.

Introduction

Urban streams begin closer to home than most people realize. During a rainstorm, every street acts as a headwater stream carrying stormwater down the drains and eventually to local lakes and streams. Like forested headwater streams, urban streets acquire substantial inputs of nutrients and other materials that feed downstream ecosystems. However, many urban streams suffer from excess inputs of nitrogen (N) and phosphorus (P), which lead to noxious algal blooms that cause lower water clarity and dissolved oxygen levels, bad odor and taste, and the loss of desirable species. Urban boulevard trees, like their streamside counterparts, are likely to be significant drivers of carbon (C), N, and P movement from land to water, through processes such as litter inputs, canopy nutrient exchange, and dust interception, but these processes are not currently well understood. It may be possible to design and manage urban forests so they not only improve their immediate neighborhoods, but also enhance the water quality and ecosystem service provisioning of local lakes and streams.

Urban trees are often touted as stormwater mitigation tools, based primarily on modeling studies suggesting that urban trees can significantly reduce stormwater volume and peak flows (Xiao et al. 1998, McPherson et al. 2011). Boulevard trees intercept rainfall that would otherwise reach the street, evaporating some of it from their leaves and diverting some down their stems and trunks (stemflow) to the soil at their base. Reducing stormflow volumes, however, does not necessarily lead to reductions in N and P inputs to stormwater, particularly if input rates of tree litter and other materials are high. To our knowledge, there have been no empirical studies quantifying urban tree effects on N or P loading to stormwater, which is not a simple function of their hydrologic effects but also includes litterfall inputs, canopy nutrient exchange, dust interception, and other processes. Although

several studies have quantified P pools and/or accumulation rates on streets, most have either avoided sampling during autumn litterfall (Selbig and Bannerman 2007)or removed leaves from samples (Waschbush et al. 1999). Sorenson and coauthors (2013) did capture autumn litterfall in their samples, but did not estimate the proportion of P loading due to tree litter inputs.

At first, it might seem straightforward to calculate boulevard trees' litter inputs to stormwater from existing data: urban forests are frequently inventoried and mapped (e.g. Kilberg and Bauer 2011), urban-specific allometric equations have been developed for estimating tree leaf biomass from basic size parameters (Nowak 1996), and tree trait data such as leaf N and P concentrations are cataloged for many species (Wright et al. 2004). However, such an approach omits many important processes that are likely to have substantial effects on N and P loading. Litter blows around extensively in urban areas, and litter inputs do not simply equal the amount of leaves overhanging streets. Urban trees exhibit wide variation in tissue nutrient contents within species (Nidzgorski et al., *in preparation*), the drivers of which are not well understood, though there is evidence that trees near roadways have higher foliar N concentrations (Bettez et al. 2013). The nutrient contents of flowers, seeds, and other non-leaf tissues are not widely cataloged, nor are there allometric equations for estimating their mass, yet these nutrient-rich organs are likely important components of N and P inputs. Even less is known about soluble N and P leaching out of tree litter, which correlate poorly with total N and P (Dorney 1986, Hobbie et al. 2013) and may be transported more readily than the litter itself along street gutters and through stormwater infrastructure.

In this study, we characterized the amounts, forms, and timing of tree litter and other material in the street gutters under different species of boulevard trees. While the entire street surface collects material that can contribute to stormwater loading, we focused our sampling on the street gutters (the portion of the street closest to the curb), which was substantially easier to sample and where we observed the bulk of material accumulating. Quantifying trees' effects on nutrient loading to stormwater is a key step towards understanding nutrient cycling in urban waters, as well as improving the design and management of urban forests to improve not only the immediate areas around the trees but

also downstream ecosystems.

We collected material from street gutters throughout the 2012 growing season under four different species of common boulevard trees, as well as streets without tree cover, and measured both bulk and soluble nutrient contents. Our specific objectives were to: 1) compare bulk and soluble N and P loading under tree species with different litter chemistry, and 2) compare seasonal patterns under trees with different phenology. For our first objective, we hypothesized that variation in tree species' litter chemistry would play a substantial role in driving differences among species' N and P loading to street gutters, and hence their potential loading to stormwater. Previous work has shown substantial differences among species in the proportion of N and P that is readily soluble (Dorney 1986, Hobbie et al. 2013), so we expected to see different patterns among species for bulk vs. soluble nutrient loading.

For our second objective, we hypothesized that differences in tree phenology would lead to markedly different seasonal patterns of N and P loading under different species. Differences in the timing and duration of litterfall, not only of leaves but also of other plant organs, could have strong interactions with management practices such as street sweeping that lead to different amounts of litter being removed vs. washed down the storm drains.

Methods

Study site

We collected samples during April-November 2012 in a residential neighborhood in western Saint Paul, Minnesota, USA. In Saint Paul and many other cities in the region, multiple contiguous city blocks are planted with a single dominant boulevard tree. We selected two city blocks (each ~200 m long) dominated by each of four species: *Acer platanoides*, *Fraxinus pennsylvanica*, *Quercus palustris*, and *Tilia cordata*, as well as two blocks without tree cover overhanging the street. Most boulevard trees on our study blocks were mature trees; average sizes are given in Table 3.1. Except for one side of one block with *F. pennsylvanica*, and one block without tree cover, there were few driveways in our sample blocks (garages are accessed via alleyways along the rear of the houses). Street parking was

common in this neighborhood, so material in the street gutters was driven over regularly. Municipal street sweeping on these residential streets took place once in the spring and once in the autumn of each year; in 2012, they were swept on April 3rd and November 15th.

Saint Paul has a humid continental climate, with average annual precipitation of 780mm (140mm as snow) and average daily high and low temperatures ranging from -4.6 and -13.6°C in January to 28.6 and 17.8°C in July (1981-2010 averages, from NOAA). Daily precipitation data for our study period were obtained from the University of Minnesota weather station, approximately 2 km away.

Sample collection

We hand-swept material from the street gutter approximately biweekly from April 1st to November 18th, 2012. We were not able to sample during or shortly after rain events, as it was not possible to sweep up fine dust if the material was wet. On each sampling date, we sampled three randomly selected gutter segments in each block (six segments total of each canopy cover) that were next to boulevards with turfgrass understory, avoiding those next to mulch, gardens, pavers, etc. or underneath trees other than the dominant species. We did not exclude gutter segments that had been sampled previously, as material in the street gutters blew around extensively and there was no apparent visual difference between segments that had or had not been sampled previously. Streets in our study neighborhood were completely resurfaced and recurbed with new stormwater piping in 2006, and the curb and 60 cm of street gutter was a single piece of fairly smooth concrete, constructed in segments averaging 2.75 m long. The smooth concrete gutter, and the lack of a joint or crack where the gutter met the curb, greatly facilitated collecting all material. We limited our sampling to the gutter because it was not possible to similarly sweep the rough asphalt of the street. Because the gutter segments varied in length, we measured the dimensions of each to convert values to grams of material per linear meter of curb (along the axis of the street).

To collect a sample, we used a dustpan and hand broom to sweep up all material including fine dust, sweeping repeatedly until we were no longer able to gather together a visible pile of dust. For large samples, some coarse material was placed into large paper bags; all fine material and the entirety of smaller samples were placed in zipper-closure plastic

bags. Samples were transported back to lab and air-dried before sieving and analysis.

Processing and leaching

We sieved air-dried samples into coarse and fine fractions using a 2mm soil sieve. We removed rocks and sticks from the coarse fraction, and weighed the coarse and fine fractions. The coarse fraction was entirely organic material, while the fine fraction was a mix of organic and mineral material. We leached and analyzed coarse and fine fractions separately for total and soluble C, N, and P.

As an index of soluble nutrients, we leached 10 g of air-dried, sieved material in 1 L of deionized water in 2.5 L plastic buckets or 1 L wide-mouth high-density polyethelene bottles. If there was less than 10 g of material available, we reduced the volume of leaching water to maintain approximately the same ratio of material to water, and recorded the exact mass and volume used. Samples were stirred or shaken by hand for 10 s (to shake bottles, we held capped bottles in a horizontal position and agitated back and forth) and allowed to sit at 22 °C for 30 min until sampling, when they were stirred or shaken briefly (2-3 s) to mix the contents. Triplicate 30 mL samples were syringe-filtered through pre-ashed GF/F filters into narrow-mouth high-density polyethelene bottles and frozen until analysis.

Leached coarse-fraction material was oven-dried at 60 °C before being analyzed for total C and N on a Costech ECS4010 element analyzer (Costech Analytical, Valencia, California, USA) at the University of Nebraska, Lincoln, and for total P by digestion with 10 N sulfuric acid after ashing samples at 300 °C for 30 min followed by 550 °C for 2 h (DeMott et al. 1998). Because the coarse material was leached before total elemental analyses, we added these values to the total dissolved C, N, and P measured in leachate to calculate bulk C, N, and P. Fine-fraction material could not be saved after leaching, so we analyzed a separate (un-leached) subsample for total C, N, and P using the same analytical protocols.

DOC and TDN were measured using a Shimadzu TOC Vcpn analyzer (Shimadzu Scientific Instruments, Columbia, MD). TDP was measured by persulfate digest followed by molybdate-blue colorimetry, and SRP by molybdate-blue chemistry. NO_3^--N concentrations were measured using vanadium-oxidation colorimetry, and NH_4^+-N concentrations using salicylate colorimetery.

Tree measurements

To characterize the boulevard trees in our study site, we measured all boulevard trees' height, trunk diameter, canopy width, canopy fullness, etc. and used the urban-specific software i-Tree Eco v5.0.8 (www.itreetools.org) to estimate leaf biomass, canopy area, and leaf area index (Table 3.1). For each of our four focal species, we selected five individuals and recorded phenology approximately weekly in spring and autumn, and approximately biweekly in summer. On each visit, we recorded the presence or absence of 11 different phenophases relating to flowers, fruits, and leaves using protocols and definitions from the National Phenology Network (www.usanpn.org).

Results

Precipitation

Precipitation was substantially different during the first and second halves of the 2012 growing season (Figure 3.1). April-June was a wet period, and we often observed that rainstorms had washed the street gutters clear of litter inputs and eroded soil and gravel into the gutters. These frequent rainstorms washed material down the gutter before we could collect it, so our data during this period underestimate actual nutrient inputs. In contrast, there was little precipitation in July-November. Throughout the second half of the growing season, we observed litter building up in the street gutters without being washed down the storm drains, and changes in total C, N, and P pools were driven primarily by litter inputs with relatively low losses.

Mass and types of material

On our first sampling date (April 1st, 2012), two days before the springtime municipal street sweeping, we collected considerable amounts of leaf litter that had been deposited after the autumn 2011 street sweeping and overwintered in the street gutters (Figure 3.2). After street sweeping removed the overwintered material, the mass of material we collected from the street gutters was fairly similar throughout the spring and summer (Figure 3.2), but the composition varied considerably from organic-dominated to mineral-dominated. In April

and May, collected material was mainly reproductive parts: flowers and bracts of all species, and seeds of *Acer spp.* and *F. pennsylvanica*. *F. pennsylvanica* seeds had been produced the previous year and held on the trees over winter. Large *Acer saccharinum* trees were common in front yards in our study area, and their seeds were also prominent in our collections in May; *A. platanoides* began dropping seeds in June. Sandy soil and small gravel comprised substantial fractions of our collections in June, which contained little coarse organic material. Beginning in July, *T. cordata* leaves were dropped in substantial quantities; most of the leaf area between veins had been eaten by insects (likely *Popillia japonica,* Japanese beetles) causing a "skeletonized" appearance. In the absence of large rainstorms, these leaves matted down and remained throughout the summer and fall. Grass clippings were also present sporadically in our collections.

Autumn litterfall patterns were considerably different among species (Figure 3.2). *F. pennsylvanica* senesced leaves dropped earlier than our other study species, in September and early October. *Q. palustris* displayed an extremely prolonged and varied litterfall. Acorns began dropping in late August, as did branch tips with green leaves (as a result of squirrels chewing the branch tips to gather nesting material). Senesced leaves began dropping in small quantities in mid-September, and dropped throughout the autumn and winter. Most *Q. palustris* leaves dropped after the November 15th street sweeping and accumulated over the winter; this phenology, as well as the high canopy cover of *Q. palustris*, led to the large mass of overwintered leaves we collected in the gutter in early April 2012. *A. platanoides* and *T. cordata* senesced leaves dropped primarily in October and concluded before the November 15th street sweeping. This is in contrast with the previous year, when both species' litterfall occurred unusually late, around the time of street sweeping or shortly afterwards; as a result, we also collected considerable amounts of *A. platanoides* and *T. cordata* litter in early April 2012. We collected a low amount of litter from *F. pennsylvanica* in early April 2012; this species dropped its leaves well before autumn street sweeping, and nearly all of its litterfall was successfully removed by street sweeping.

Bulk C, N, and P

Concentrations of bulk C, N, and P varied more between size fractions and over time

than among the tree canopy species (Figure 3.3a). The C content of coarse organics was relatively constant, but the C content of the fine fraction increased from low-C material during the spring and early summer, when we observed more soil in our collections, to high-C material in the fall, when the fine fraction was largely small pieces of leaf litter. Percents N and P both declined in the coarse organic fraction over the course of our sampling period, N more strongly than P. In the fine fraction, percents N and P did not exhibit the same magnitude of change between summer and fall as did percent C. Bulk C:N and C:P ratios increased over our sampling season, driven largely by increases in C, and N:P ratio was relatively constant (Figure 3.3b). Variation in the total mass of C, N, and P per meter of curb was driven more by variation in the mass of material than by variation in the percent C, N, or P (Figure 3.3c).

For our three study species with autumn pulses of litterfall (excluding *Q. palustris*, whose litterfall continued throughout the winter), autumn litterfall from these species contributed 8.3-10.4 kg N and 0.54-0.78 kg P curb-km^{-1} to the street gutters (Figure 3.3a, converting from g/m to kg/km). To estimate nutrient loading on an areal basis (kg/km^2), we scaled up to the small subwatershed where our study neighborhood is located, which has 6.2 km of curb and a watershed area of 0.235 km$^{2.}$, and is relatively uniform in terms of boulevard tree canopy cover and species composition. We estimate that autumn litterfall contributed of 219.0-274.4 kg N km^{-2} and 14.2-20.6 kg P km^{-2} to the street gutters.

Soluble C, N, and P

Soluble C, N, and P in our 30 min lab leaching measurements showed spikes on individual sampling dates more so than clear seasonal trends, whether expressed as mg leached per g of litter (Figure 4a-c) or as a percentage of bulk C, N, or P that leached out (Figure 3.4d). There were clearer seasonal trends in DOC:TDN and DOC:TDP ratios increasing over August-November, driven largely by increases in DOC concentrations. In contrast, TDN:TDP ratios did not exhibit seasonal trends (Figure 4e). The percentage of bulk C, N, or P that leached out changed differently throughout the year for different species; as a result, there were not consistent differences among species (Figure 3.4d).

C and N were less soluble than P; generally less than 5% of total C or N leached out

during our lab leaching measurements, in contrast with 10-20% or more of P. TDN was mostly composed of DON, whereas TDP was mostly composed of SRP (Figure 4f), though there were some sampling dates where the other forms dominated. In contrast with bulk C, N, and P, variation in the total mass of soluble C, N, and P in the street gutters was driven by variation in both the mass of material and its chemistry (Figure 4g). This is seen most clearly in the two October samples, where soluble mg/g litter decreased and drove a sharp drop in total soluble nutrient pools even though the total mass of material in the gutter stayed constant or increased.

Discussion

Inputs vs. transformations

Patterns in the composition of material in the street gutters were driven by changing inputs as well as transformations in and transport out of the gutter. The relative importance of these varied considerably over the course of our study, especially as material remained in the gutter much longer during the low-precipitation second half of our sampling period.

Following spring street sweeping, material in the street gutters was dominated by fresh, untransformed litter inputs. Municipal street sweeping cleared out overwintered material on April 3rd, and rainstorms frequently washed litter down the storm drains, so most of the material we collected in April and May had fallen within the previous 3-4 days. We expect that the springtime nutrient inputs to stormwater were considerably higher than our measurements suggest, as much of this N- and P-rich material was washed down the drains shortly after it fell and before our next sampling date. On the *Q. palustris* blocks on April 21st, some of the material was overwintered litter that was packed down and had not been fully removed by the April 3rd street sweepings.

In summer and autumn 2012, samples were varying mixes of fresh inputs and older litter that had undergone some decomposition. The most visible example of material building up and decomposing was the *T. cordata* blocks, where the skeletonized leaves remained in the gutter for months and formed solid mats that could be peeled up almost as a single unit. Bulk C, N, and P concentrations did not appear to undergo much change as litter remained

in the gutter (Figure 3.3), but the transient spikes in soluble C, N, and P concentrations (Figure 3.4) suggest that soluble pools were undergoing substantial transformations and losses. A striking example of this is the sharp decline in soluble C, N, and P between October 7th and 21st. The two weeks preceding October 7th had been completely dry (Figure 3.1), and litter inputs at this time were primarily senesced leaves. Between October 7-21, small rain events appear to have leached out much of the soluble C, N, and P from the material in between these two sampling dates; this is corroborated by runoff samples collected during two rain events during that period that showed high DOC, DON, and TDP concentrations on October 13th but much lower concentrations on the 19th (J. Finlay and B. Janke, *unpublished data*).

The decline in bulk N and P concentrations in the coarse organic fraction over the growing season (Figure 3.3a) is likely driven by seasonal changes in input chemistry, as inputs shifted from flowers, seeds, and young leaves in the spring, to older green leaves in summer, and finally senesced leaves in autumn. We do not expect that much of this pattern was due to losses of N and P while material remained in the street gutter. In spring and early summer, collected material was relatively fresh since frequent rainstorms washed material out of the street gutters. Late summer and autumn were relatively dry, so we expect there were minimal losses of total N and P. A similar pattern was observed in coarse organic material collected by municipal street sweeping in 2011 and 2012 in Prior Lake, MN, a suburb 35 km SW of our study site (Kalinosky et al., *in preparation*). Streets in that study were swept one, two, or four times a month, so all litter collected was relatively fresh inputs and exhibited a similar decline in N and P concentration over the season. Thus, the seasonal declines in bulk N and P concentration in this study are also likely driven by changing litter input chemistry.

Managing the urban forest for water quality

Litter cleanup efforts such as street sweeping can potentially reduce nutrient loading to stormwater, but only if litter is removed promptly before it can leach soluble nutrients, decompose, or wash down the storm drains. Removing all autumn litterfall promptly could potentially prevent 219.0-274.4 kg N km^{-2} and 14.2-20.6 kg P km^{-2} from washing down the storm drains.

However, the substantial differences we found in the timing of litterfall among our study species makes it difficult for a single autumn street sweeping to achieve maximum nutrient removal for multiple species. During our study, the autumn street sweeping came more than a month after the peak of *F. pennsylvanica* litterfall and before *Q. palustris* had fully dropped its leaves. *F. pensylvanica* autumn litterfall contributed 10.4 kg N and 0.68 kg P curb-km^{-1} (Figure 3.3a); if there had been appreciable rainfall in the month between litterfall and street sweeping, we expect that most of the *F. pennsylvanica* litter would have washed down the storm drains and increased stormwater loading by this amount. Matching litter cleanup to tree litterfall also requires flexibility to adjust to interannual variation in phenology; otherwise, this can result in such situations as occurred in 2011 when some species' litterfall was unusually late and after municipal street sweeping.

We also found substantial inputs of litter during times other than peak autumn leaf-drop, none of which are swept up by current street-sweeping practices. The springtime pulse of reproductive litter inputs also contributed high N and P loads, though most of this washed down the storm drains before our sampling dates. Stormwater N and P loading data for the Capitol Region Watershed showed higher N and P loading during spring than autumn (B. Janke, *unpublished data,* 2007-2012 average), underscoring the potential contributions of springtime litter inputs. The summertime drop of skeletonized *T. cordata* leaves contributed 1.8 kg N and 0.12 kg P curb-km^{-1} at its peak in late July, which in a wetter summer would likely have washed down the drains during intense summer rainstorms.

We recommend that cities explore options to decrease the time lag between litterfall and litter cleanup, as well as to target non-autumn litterfall, and compare the costs and benefits of using this strategy to reduce stormwater nutrient loads. In our study city of Saint Paul, we expect that municipal street sweeping has limited flexibility to respond to individual species' phenology, since the city has over 1,400 km of streets to sweep. However, individual and neighborhood-level residents' efforts for litter cleanup may have more ability to match tree litter cleanup to litterfall timing. In the neighborhood including our study area, the Como Lake Neighbors Network (www.clnn.org) has organized an annual grassroots effort to motivate and assist residents in raking up leaves from their street gutters each autumn. In 2012 and prior years, they selected a single weekend for raking throughout the

neighborhood, but in 2013, in response to our preliminary findings, they adopted a new model where each block self-organized to rake up litter shortly after the dominant boulevard species' litterfall. This flexibility to clean up litter at different times on different blocks, and to adapt to interannual variability, gives grassroots efforts the potential to make substantial contributions to reducing N and P loading to stormwater.

Species selection

Just as street sweeping must be designed to account for differences in phenology, selecting boulevard tree species must also be sensitive to management considerations. We had hypothesized that tree litter chemistry would drive substantial differences among species' nutrient inputs to stormwater, but instead we found that phenology is a much more important driver of differences among species. As a result, we do not recommend shifting the planting mix of boulevard trees to favor those with lower N and P concentrations in their litter, as this would not be likely to yield significant reductions in stormwater nutrient inputs.

Instead, we recommend selecting boulevard tree species whose litterfall timing is amenable to being cleaned up soon thereafter, either with current or improved litter cleanup practices. Saint Paul Parks & Recreation is working to increase the species diversity of the city's urban forest, and has developed new planting guidelines that call for moving away from long stretches of a single boulevard tree species and interplanting two or more species within city blocks (Saint Paul Department of Parks and Recreation 2010). They recommend interplanting species that have compatible aesthetics and pruning schedules, and our study highlights the importance of also considering the phenology of different litterfall. Improving municipal street sweeping or grassroots efforts such that litter cleanup matches litterfall timing is more likely to succeed if all trees on a given city block drop their litter at approximately the same time and can be cleaned up at once.

Some boulevard tree species are badly mismatched to certain management practices and are contraindicated from a water-quality perspective. *Q. palustris* had a very protracted autumn litterfall, starting to drop branch tips bearing green leaves in early September, and continuing to drop senesced leaves slowly throughout the entire winter. Although this would

be less problematic in areas with frequent street sweeping that continues late into the fall and resumes early in the spring, in our study area, which is swept twice per year, this led to large buildups of leaf litter overwintering in the street gutters (Figure 3.3). While much of this overwintered leaf pack remained in the street gutters in April 2012 and was removed by spring street sweeping, this was not the case in April-May 2013 when a heavy snowmelt and heavy rains washed most of the leaf pack down the storm gutters before the spring street sweeping (*personal observations*). Litterfall from red oaks such as *Q. palustris* cannot be swept up with a single autumn street sweeping, and from a water-quality perspective are not a good species choice for areas with infrequent street sweeping.

Table 3.1: Average size of boulevard trees in our study area, by species. Diameter at breast height (DBH) and tree height were measured empirically; all other values were calculated using the urban-specific software i-Tree Eco v5.0.8 (www.itreetools.org)

Species Name	DBH (cm)	Tree height (m)	Projected canopy area (m2)	Leaf area (m2)	Leaf biomass (kg)	Leaf area index
Tilia cordata	31.92	10.92	55.23	377.85	17.57	7.15
Fraxinus pennsylvanica	40.90	14.45	93.57	413.38	26.96	4.69
Acer platanoides	31.25	10.86	68.77	398.22	21.49	6.18
Quercus palustris	46.81	16.41	116.50	339.51	30.35	3.01

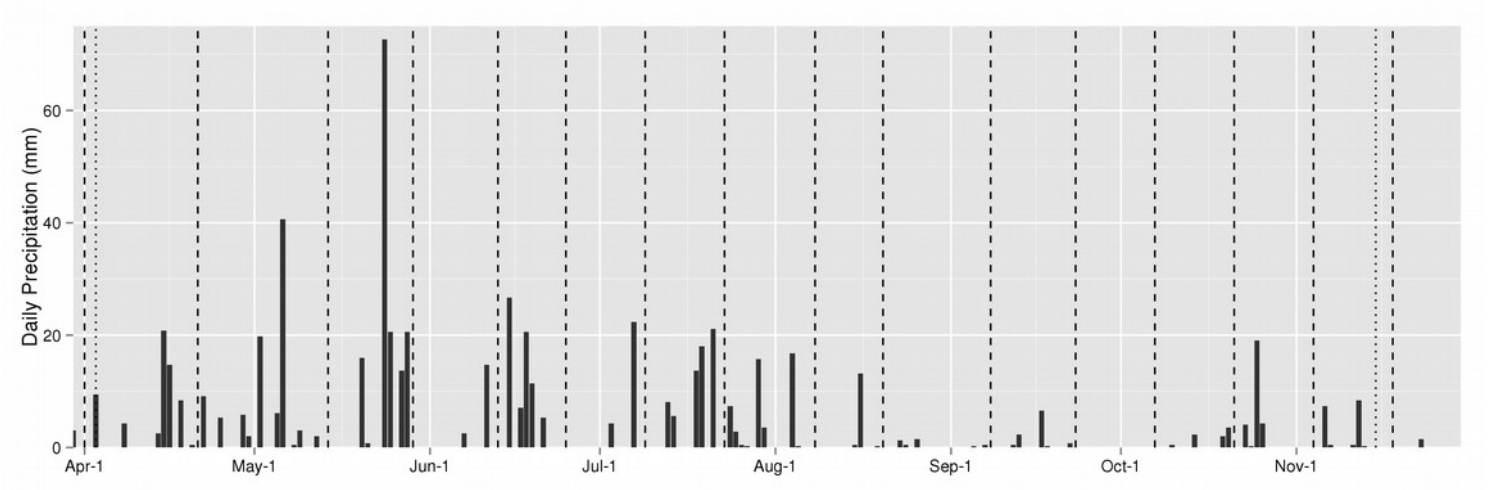


Figure 3.1: Daily precipitation (mm) for April-November 2012. Vertical dashed lines indicate sampling dates, and vertical dotted lines indicate municipal street sweeping dates (April 3st and November 15th)

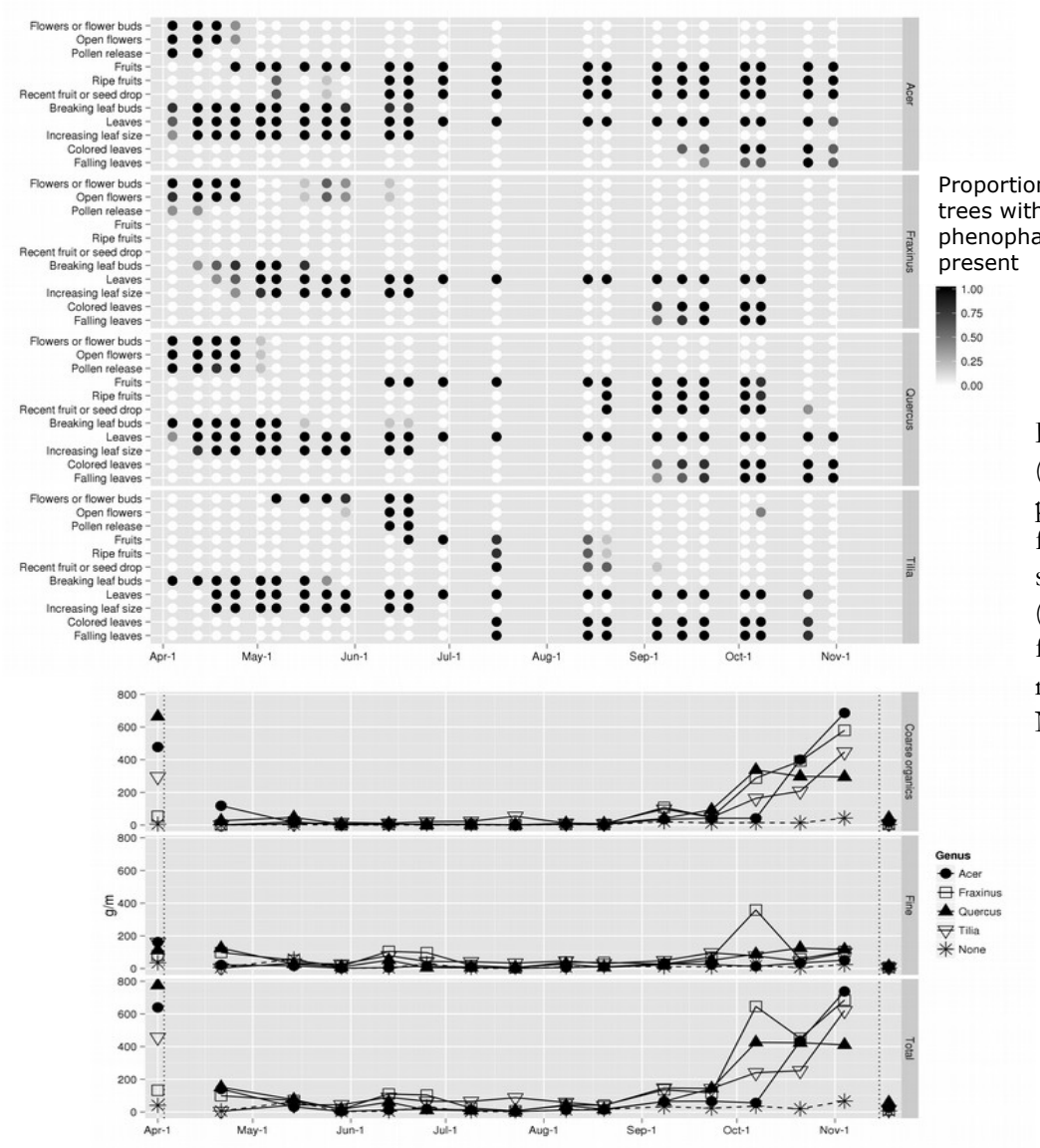
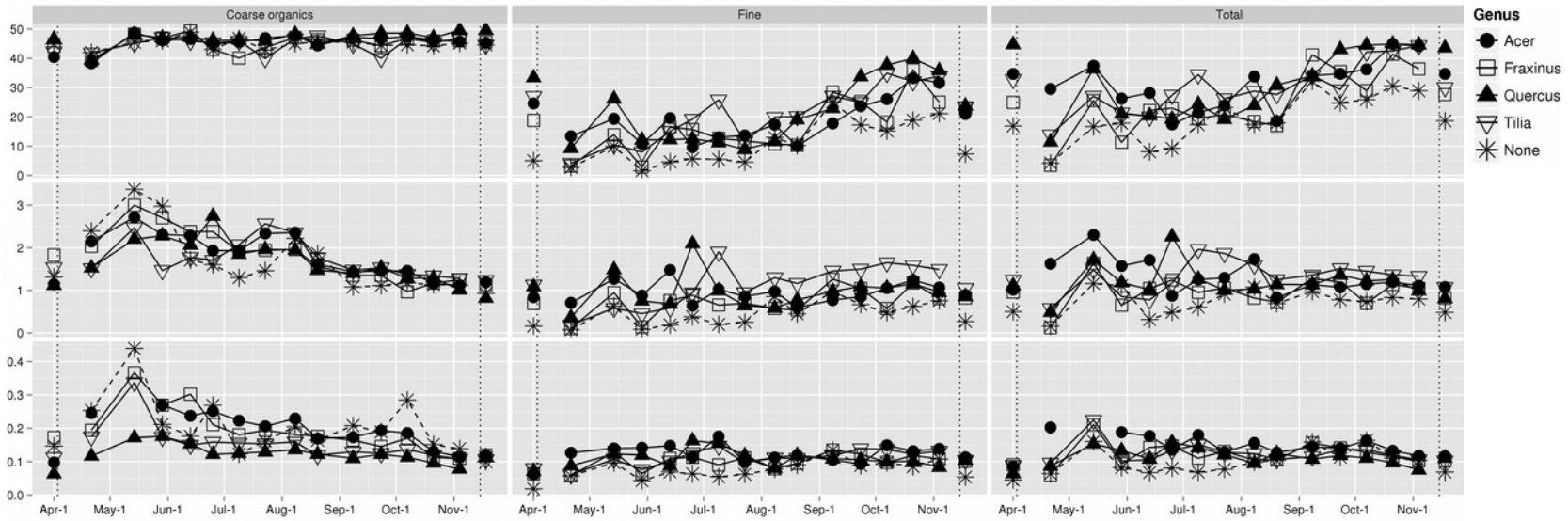
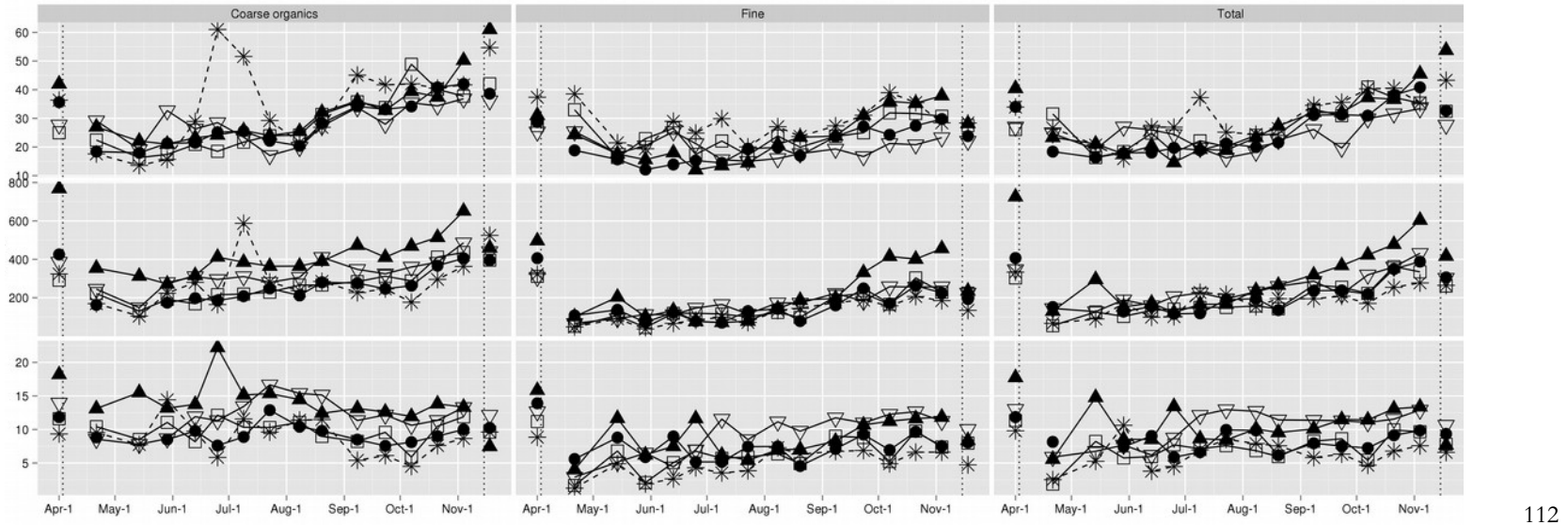


Figure 3.2: Sources and amounts of tree litter inputs. (a) Phenology records. Point color indicates the proportion of individual trees (n=5 for each species) for which a given phenophase was present on each sampling date.
(b) Average mass (g/m) of coarse organic fraction, fine fraction, and total. Vertical dotted lines indicate municipal street sweeping dates (April 3st and November 15th).

3a:


Coarse organics
Fine
Total
%C
%N
%P
Genus
Acer
Fraxinus
Quercus
Tilia
None
Apr-1
May-1
Jun-1
Jul-1
Aug-1
Sep-1
Oct-1
Nov-1

3b:


Coarse organics
Fine
Total
C:N
C:P
N:P
Apr-1
May-1
Jun-1
Jul-1
Aug-1
Sep-1
Oct-1
Nov-1

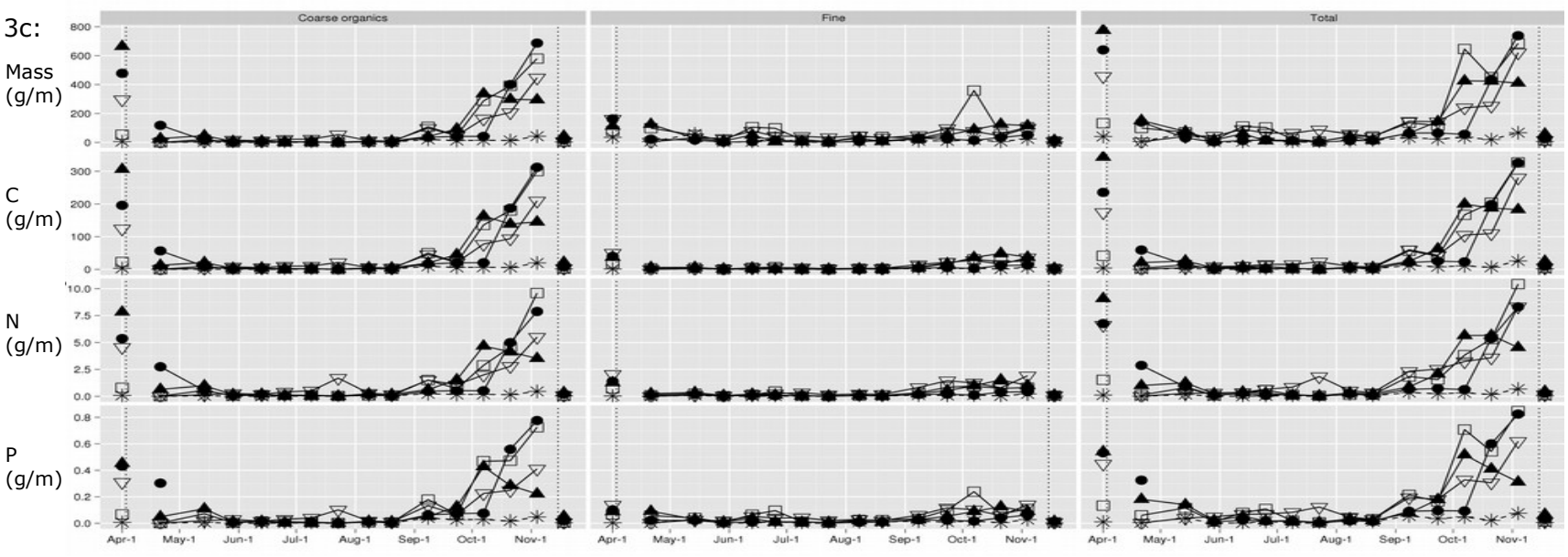
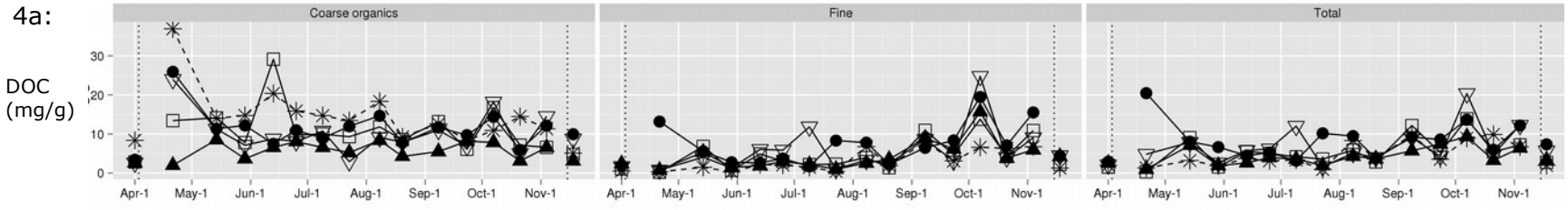
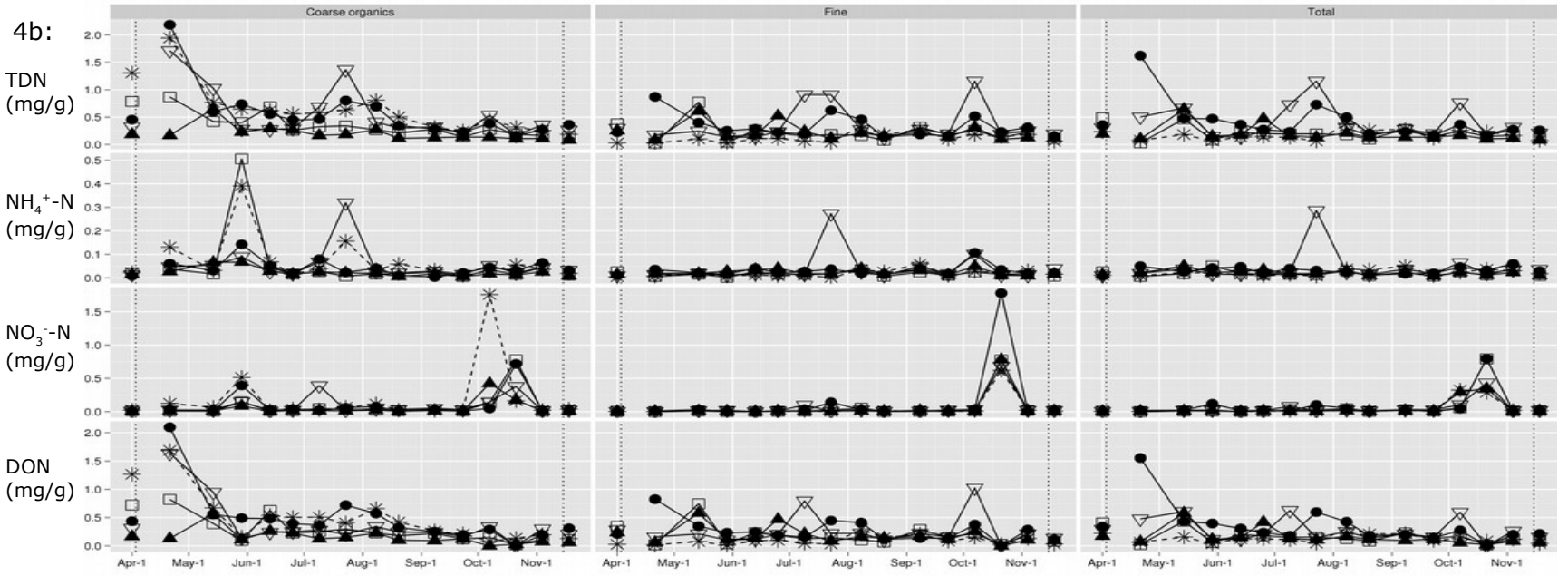


Figure 3.3: Bulk C, N, and P in collected material. Graphs are faceted across by fraction (coarse organic, fine, and total) and down by analyte or measurement. Vertical dotted lines indicate municipal street sweeping dates (April 3st and November 15th).
(a) Percent C, N, and P.
(b) C:N, C:P, and N:P ratios (by mass).
(c) Total mass (g/m) of C, N, and P; total mass is also repeated here to facilitate comparison.

4a:

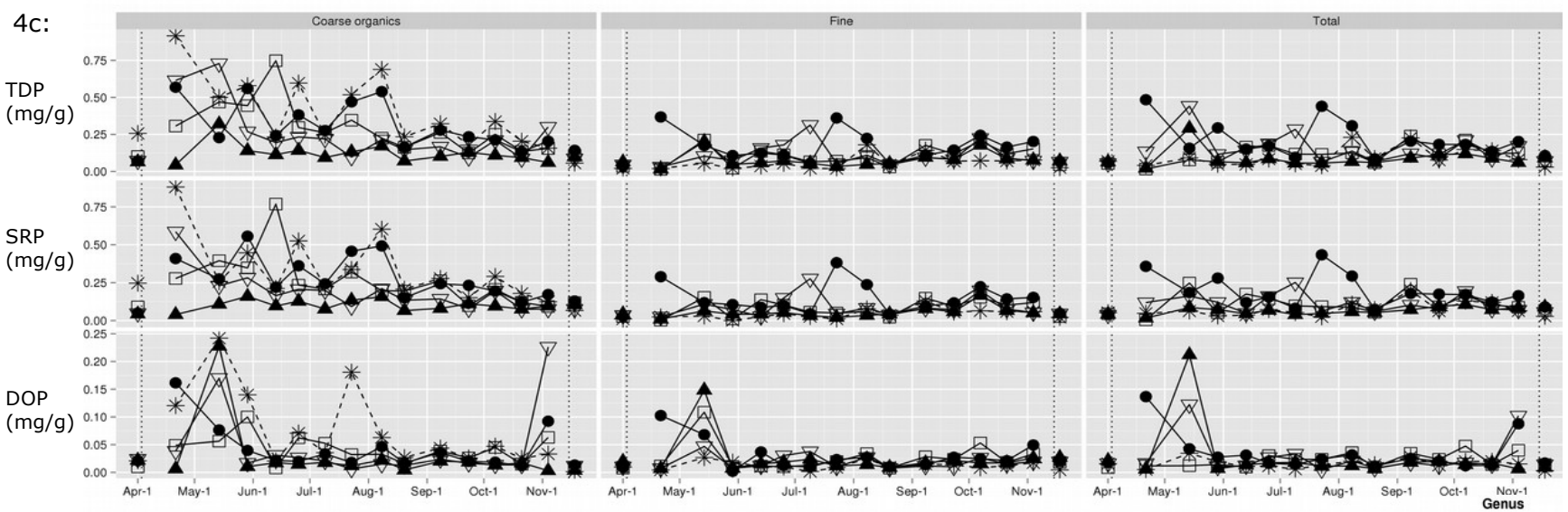

Coarse organics
Fine
Total
DOC (mg/g)
0
10
20
30
Apr-1
May-1
Jun-1
Jul-1
Aug-1
Sep-1
Oct-1
Nov-1

4b:

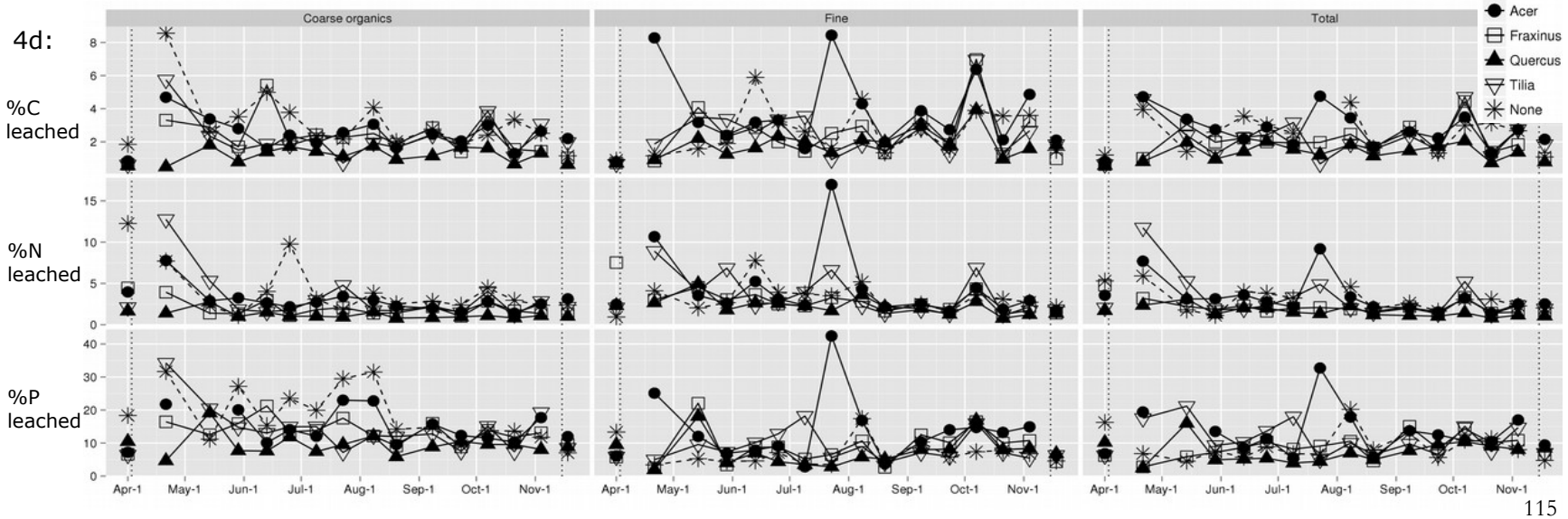

Coarse organics
Fine
Total
TDN (mg/g)
NH4+-N (mg/g)
NO3--N (mg/g)
DON (mg/g)
Apr-1
May-1
Jun-1
Jul-1
Aug-1
Sep-1
Oct-1
Nov-1


Genus
Acer
Fraxinus
Quercus
Tilia
None

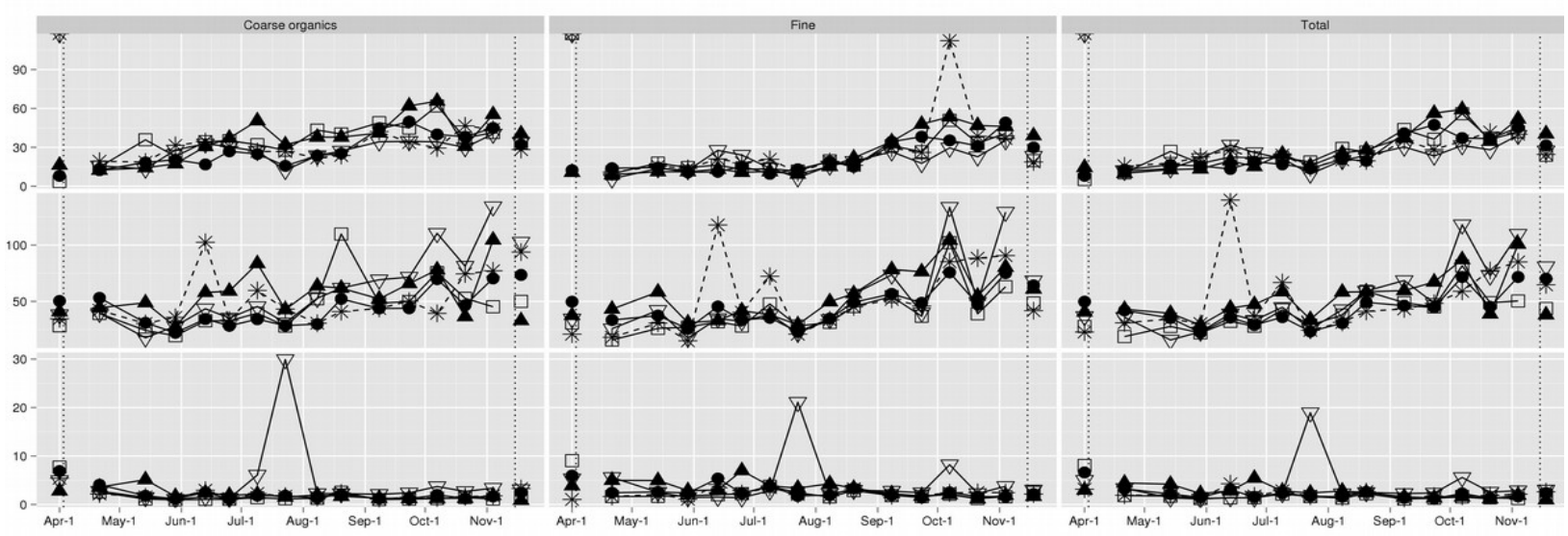
4c:


Coarse organics
Fine
Total
TDP (mg/g)
SRP (mg/g)
DOP (mg/g)
Apr-1
May-1
Jun-1
Jul-1
Aug-1
Sep-1
Oct-1
Nov-1

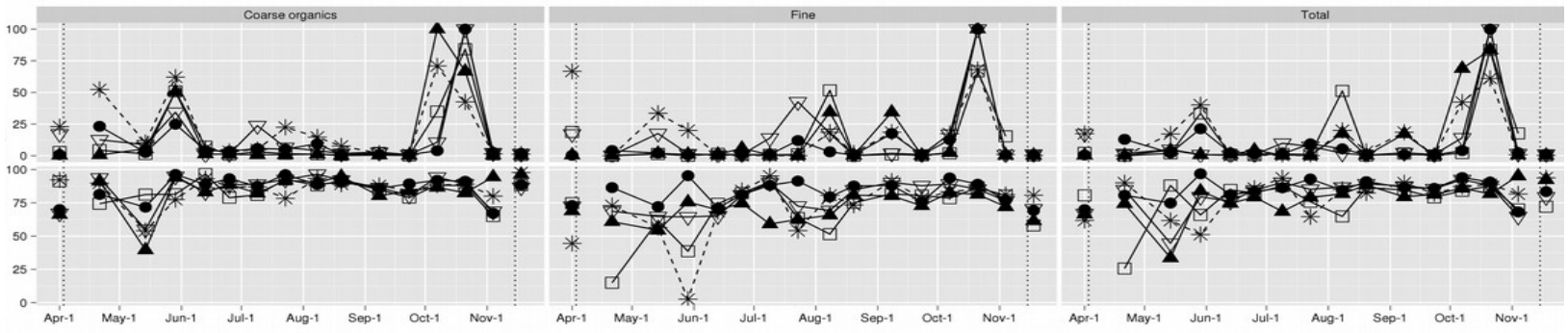
4d:


Coarse organics
Fine
Total
%C leached
%N leached
%P leached
Genus
Acer
Fraxinus
Quercus
Tilia
None
Apr-1
May-1
Jun-1
Jul-1
Aug-1
Sep-1
Oct-1
Nov-1

4e:


Coarse organics
Fine
Total
DOC: TDN
DOC: TDP
TDN: TDP
Apr-1
May-1
Jun-1
Jul-1
Aug-1
Sep-1
Oct-1
Nov-1

4f:


Coarse organics
Fine
Total
%TDN as inorganic
%TDP as inorganic
Apr-1
May-1
Jun-1
Jul-1
Aug-1
Sep-1
Oct-1
Nov-1


Genus
Acer
Fraxinus
Quercus
Tilia
None

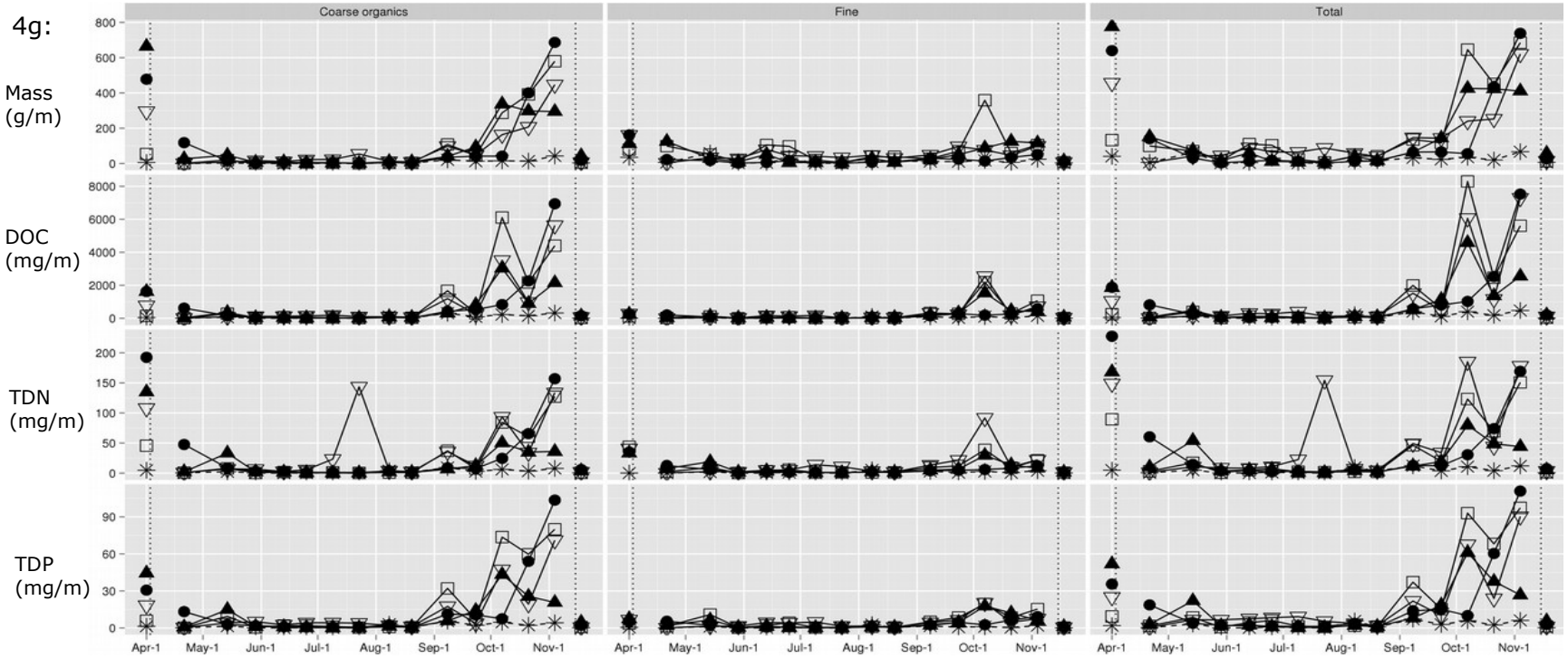


Figure 3.4: Soluble C, N, and P in collected material. Graphs are faceted across by fraction (coarse organic, fine, and total) and down by analyte or measurement. Vertical dotted lines indicate municipal street sweeping dates (April 3st and November 15th).
(a) DOC concentrations (mg/g litter).
(b) Soluble N concentrations (mg/g litter), as: TDN, NH_4^+, NO_3^-, and DON.
(c) Soluble P concentrations (mg/g litter), as: TDP, SRP, DOP.
(d) Percent of bulk C, N, and P that leached out in 30 min laboratory leaching.
(e) DOC:TDN, DOC:TDP, and TDN:TDP ratios (by mass).
(f) Percent of TDN and TDP as inorganic forms.
(g) Total mass (mg/m) of DOC, TDN, and TDP; total mass (g/m) is also repeated here to facilitate comparison.

Conclusion

Although improving waste management has the potential to substantially increase nutrient recycling, this approach offers little opportunity for reducing nutrient pollution (Chapter 1). However, this does not mean that reducing nutrient consumption is the only tool available for reducing nutrient pollution. Urban nutrient fluxes move not only through anthropogenic waste streams, but also through the biophysical environment including vegetation and soil. As shown in Chapters 2 and 3, urban trees are substantial drivers of N and P transport and retention between land and water. Using these results to design and manage the urban forest for water quality has the potential to complement source-reduction efforts in reducing N and P pollution

Urban trees are drivers of both nutrient transport and nutrient retention. Autumn 2012 leaf-litter inputs added 219-274 kg N and 14-21 kg P per km^2 to street gutters (range is across different species in our study). In comparison, urban trees across the Capitol Region Watershed retained 3.1 kg P/km^2 in 2012 and 6.9 kg P/km^2 in 2013, roughly 20-30% of autumn leaf-litter inputs. For N, urban trees are net sources of N moving from land to water, if our study years are representative of N leaching. For P, by contrast, trees' net effect depends on how litterfall is managed. If litter inputs to storm gutters were removed promptly by street sweeping, trees' contributions to stormwater loading could potentially be reduced below the amount of P they retain from leaching to groundwater. If much of the P in litter inputs were allowed to wash down the storm drains in particulate or dissolved forms, however, trees would be a substantial net source of P as well.

Our results suggest that planting more trees in yards, parks, and other areas away from roads and other impervious surfaces would reduce P leaching to groundwater without also increasing nutrient loading to stormwater. While planting fewer boulevard trees (or selecting species with less leaf biomass and hence lower litter inputs) would reduce N and P inputs to stormwater, doing so would also reduce tree-provided ecosystem services such as: shading buildings, roads, and other surfaces (which reduces the heat-island effect and energy demands for summertime cooling), removing air pollutants, increasing evapotranspiration (which reduces stormwater peak flows and also increases summertime evaporative cooling),

slowing traffic, and contributing to neighborhood aesthetics and social well-being. Designing the future urban forest must weigh the tradeoffs among ecosystem services and disservices, and also take into consideration the effects of management actions such as street sweeping.

Prompt tree litter cleanup could not only reduce nutrient pollution but also increase nutrient recycling. We estimated that composting leaves from residential yards could return one-third as much P to gardens and croplands as households consume in food (Chapter 1), but these calculations excluded all litter from boulevard trees, since the boulevard is City property and not part of the residential parcel. Collecting boulevard trees' litter from yards and streets could further increase nutrient recycling; Saint Paul already screens the material from autumn street sweeping to separate leaf litter from rocks and debris, and composts most of the leaf litter to reduce landfill costs. Adjusting the timing of street sweeping to increase nutrient collection would increase the amounts of N and P composted, without the need for additional changes. The combined benefits of reducing nutrient pollution and increasing nutrient recycling might help make a compelling case to optimize street sweeping for nutrient collection.

Overall, our findings are hopeful: the Twin Cities metropolitan area offers a number of opportunities to increase nutrient sustainability by changing the ways we manage anthropogenic waste streams and the urban forest. Moving forward, it will be necessary to draw on findings like these, which examine incremental changes in isolated components of the urban environment, and synthesize them into a systems-level understanding of the urban socio-ecological system that considers many of the interactions and feedbacks. This will make it possible to consider a more complete set of benefits and costs for any possible solution, as well as to address questions of vulnerability and resilience that only emerge at the systems level. It is also important to consider not only incremental changes but also solutions that would fundamentally re-envision and transform the entire urban system. As we learn how human actions and choices interact with climatic, biotic, and geochemical processes to drive urban nutrient cycling, it will become increasingly possible to design and manage urban ecosystems in ways that move towards a sustainable, resilient future.

Bibliography

Amador, J., R. Hull, E. Patenaude, J. Bushoven, and J. Gorres. 2007. Potential nitrate leaching under common landscaping plants. Water Air and Soil Pollution 185:323–333.

Anoka County. 2012. Anoka County solid waste management master plan.

Baker, L. A., P. M. Hartzheim, S. E. Hobbie, J. Y. King, and K. C. Nelson. 2007. Effect of consumption choices on fluxes of carbon, nitrogen and phosphorus through households. Urban Ecosystems 10:97–117.

Banks, C. J., M. Chesshire, S. Heaven, and R. Arnold. 2011. Anaerobic digestion of source-segregated domestic food waste: Performance assessment by mass and energy balance. Bioresource Technology 102:612–620.

Barzen, M., R. Piva, C. Y. Wu, and R. Dahlman. 2008. Residential fuelwood assessment, State of Minnesota, 2007-2008 heating season. Minnesota Department of Natural Resources.

Beauchemin, S., R. R. Simard, and D. Cluis. 1998. Forms and Concentration of Phosphorus in Drainage Water of Twenty-Seven Tile-Drained Soils. Journal of Environmental Quality 27:721.

Beck, R. W. 2000. Statewide MSW composition study: A study of discards in the state of Minnesota. Solid Waste Management Coordinating Board, Saint Paul, MN.

Bettez, N. D., R. Marino, R. W. Howarth, and E. A. Davidson. 2013. Roads as nitrogen deposition hot spots. Biogeochemistry 114:149–163.

Boyden, S., S. Millar, K. Newcombe, and B. O'Neill. 1981. The Ecology of a city and its people: The case of Hong Kong. Australian National University.

Bray, R. H., and L. T. Kurtz. 1945. Determination of total, organic, and available forms of phosphorus in soils. Soil science 59:39–46.

Burkart, M. R., W. W. Simpkins, A. J. Morrow, and J. M. Gannon. 2004. Occurrence of Total Dissolved Phosphorus in Unconsolidated Aquifers and Aquitards in Iowa. Journal of the American Water Resources Association 40:827–834.

Capitol Region Watershed District. 2010. Stormwater BMP performance assessment and cost-benefit analysis. Page 107. Saint Paul, MN.

Ceriani, L., and P. Verme. 2012. The origins of the Gini index: extracts from Variabilità e Mutabilità (1912) by Corrado Gini. The Journal of Economic Inequality 10:421–443.

Christensen, A., R. Westerholm, and J. Almén. 2001. Measurement of Regulated and Unregulated Exhaust Emissions from a Lawn Mower with and without an Oxidizing Catalyst: A Comparison of Two Different Fuels. Environmental Science & Technology 35:2166–2170.

Clark, C. M., S. E. Hobbie, R. Venterea, and D. Tilman. 2009. Long-lasting effects on nitrogen cycling 12 years after treatments cease despite minimal long-term nitrogen retention. Global Change Biology 15:1755–1766.

Cordell, D., A. Rosemarin, J. J. Schröder, and A. L. Smit. 2011. Towards global phosphorus security: A systems framework for phosphorus recovery and reuse options. Chemosphere 84:747–758.

Cordell, D., and S. White. 2013. Sustainable Phosphorus Measures: Strategies and Technologies for Achieving Phosphorus Security. Agronomy 3:86–116.

Crow, P. 2005. The Influence of Soils and Species on Tree Root Depth. Forestry

Commission Information Note.
Dahmus, M. E., and K. C. Nelson. 2014. Yard stories: examining residents' conceptions of their yards as part of the urban ecosystem in Minnesota. Urban Ecosystems 17:173–194.
DeMott, W. R., R. D. Gulati, and K. Siewertsen. 1998. Effects of phosphorus-deficient diets on the carbon and phosphorus balance of Daphnia magna. Limnology and Oceanography 43:1147–1161.
Dewalle, D. R., and G. M. Heisler. 2012. Use of Windbreaks for Home Energy. Agriculture, Ecosystems, and the Environment 22:243–260.
Diak, J., B. Örmeci, and K. J. Kennedy. 2012. Effect of enzymes on anaerobic digestion of primary sludge and septic tank performance. Bioprocess and Biosystems Engineering 35:1577–1589.
Dijkstra, F. A., K. Wrage, S. E. Hobbie, and P. B. Reich. 2006. Tree Patches Show Greater N Losses but Maintain Higher Soil N Availability than Grassland Patches in a Frequently Burned Oak Savanna. Ecosystems 9:441–452.
Djodjic, F., K. Börling, and L. Bergström. 2004. Phosphorus leaching in relation to soil type and soil phosphorus content. Journal of Environmental Quality 33:678–684.
Dorney, J. R. 1986. Leachable and total phosphorous in urban street tree leaves. Water, Air, and Soil Pollution 28:439–443.
Elser, J. J., M. E. S. Bracken, E. E. Cleland, D. S. Gruner, W. S. Harpole, H. Hillebrand, J. T. Ngai, E. W. Seabloom, J. B. Shurin, and J. E. Smith. 2007. Global analysis of nitrogen and phosphorus limitation of primary producers in freshwater, marine and terrestrial ecosystems. Ecology Letters 10:1135–1142.
EPA Science Advisory Board. 2007. Hypoxia in the Northern Gulf of Mexico: An update by the EPA Science Advisory Board. Page 333. US Environmental Protection Agency, Washington, D.C.
Erickson, J. E., J. L. Cisar, G. H. Snyder, and J. C. Volin. 2005. Phosphorus and Potassium Leaching under Contrasting Residential Landscape Models Established on a Sandy Soil. Crop Science 45:546.
Erickson, J. E., J. L. Cisar, G. H. Snyder, D. M. Park, and K. E. Williams. 2008. Does a Mixed-Species Landscape Reduce Inorganic-Nitrogen Leaching Compared to a Conventional St. Augustinegrass Lawn? Crop Science 48:1586.
Faerge, J., J. Magid, and F. W. Penning de Vries. 2001. Urban nutrient balance for Bangkok. Ecological modelling 139:63–74.
Federer, C. A., C. Vorosmarty, and B. Fekete. 2003. Sensitivity of annual evaporation to soil and root properties in two models of contrasting complexity. Journal of Hydrometeorology 4.
Fissore, C., L. A. Baker, S. E. Hobbie, J. Y. King, J. P. McFadden, K. C. Nelson, and I. Jakobsdottir. 2011. Carbon, nitrogen, and phosphorus fluxes in household ecosystems in the Minneapolis-Saint Paul, Minnesota, urban region. Ecological Applications 21:619–639.
Fissore, C., S. E. Hobbie, J. Y. King, J. P. McFadden, K. C. Nelson, and L. A. Baker. 2012. The residential landscape: fluxes of elements and the role of household decisions. Urban Ecosystems 15:1–18.
Geiger, S. C., R. J. Vandenbeldt, and A. Manu. 1994. Variability in the growth of Faidherbia

albida: the soils connection. Soil Science Society of America journal (USA).
Gill, R. A., and I. C. Burke. 1999. Ecosystem consequences of plant life form changes at three sites in the semiarid United States. Oecologia 121:551–563.
Gordon, H., P. M. Haygarth, and R. D. Bardgett. 2008. Drying and rewetting effects on soil microbial community composition and nutrient leaching. Soil Biology and Biochemistry 40:302–311.
Greenan, C. M., T. B. Moorman, T. C. Kaspar, T. B. Parkin, and D. B. Jaynes. 2006. Comparing Carbon Substrates for Denitrification of Subsurface Drainage Water. Journal of Environment Quality 35:824.
Grimm, N. B., S. H. Faeth, N. E. Golubiewski, C. L. Redman, J. Wu, X. Bai, and J. M. Briggs. 2008. Global Change and the Ecology of Cities. Science 319:756–760.
Groffman, P., C. Williams, R. Pouyat, L. Band, and I. Yesilonis. 2009. Nitrate Leaching and Nitrous Oxide Flux in Urban Forests and Grasslands. Journal of Environmental Quality 38:1848–1860.
Gundersen, P., I. Callesen, and W. de Vries. 1998. Nitrate leaching in forest ecosystems is related to forest floor C/N ratios. Environmental Pollution 102:403–407.
Hammer, R. B., S. I. Stewart, R. L. Winkler, V. C. Radeloff, and P. R. Voss. 2004. Characterizing dynamic spatial and temporal residential density patterns from 1940–1990 across the North Central United States. Landscape and Urban Planning 69:183–199.
Handcock, M. S. 2013. Relative Distribution Methods. Project home page at http://www.stat.ucla.edu/~handcock/RelDist.
Harpole, W. S., J. T. Ngai, E. E. Cleland, E. W. Seabloom, E. T. Borer, M. E. S. Bracken, J. J. Elser, D. S. Gruner, H. Hillebrand, J. B. Shurin, and J. E. Smith. 2011. Nutrient co-limitation of primary producer communities. Ecology Letters 14:852–862.
Haygarth, P. M., L. Hepworth, and S. C. Jarvis. 1998. Forms of phosphorus transfer in hydrological pathways from soil under grazed grassland. European Journal of Soil Science 49:65–72.
Hobbie, S. E. 1992. Effects of plant species on nutrient cycling. Trends in Ecology & Evolution 7:336–339.
Hobbie, S. E., L. A. Baker, C. Buyarski, D. Nidzgorski, and J. C. Finlay. 2013. Decomposition of tree leaf litter on pavement: implications for urban water quality. Urban Ecosystems:1–17.
Hooda, P. S., M. Moynagh, and I. F. Svoboda. 1999. Phosphorus loss in drainflow from intensively managed grassland soils. Journal of Environmental Quality 28:1235–42.
Howarth, R., F. Chan, D. J. Conley, J. Garnier, S. C. Doney, R. Marino, and G. Billen. 2011. Coupled biogeochemical cycles: eutrophication and hypoxia in temperate estuaries and coastal marine ecosystems. Frontiers in Ecology and the Environment 9:18–26.
Jackson, R. B., J. Canadell, J. R. Ehleringer, H. A. Mooney, O. E. Sala, and E. D. Schulze. 1996. A Global Analysis of Root Distributions for Terrestrial Biomes. Oecologia 108:389–411.
Jackson, R. B., J. L. Banner, E. G. Jobbagy, W. T. Pockman, and D. H. Wall. 2002. Ecosystem carbon loss with woody plant invasion of grasslands. Nature 418:623–626.
Janke, B. D., J. C. Finlay, S. E. Hobbie, L. A. Baker, R. W. Sterner, D. Nidzgorski, and B. N. Wilson. 2013. Contrasting influences of stormflow and baseflow pathways on

nitrogen and phosphorus export from an urban watershed. Biogeochemistry:1–20.
Kaye, J. P., P. M. Groffman, N. B. Grimm, L. A. Baker, and R. V. Pouyat. 2006. A distinct urban biogeochemistry? Trends in Ecology & Evolution 21:192–199.
Kilberg, D., and M. E. Bauer. 2011, October 16. Mapping Capitol Region Watershed District Urban Tree Canopy.
Koenker, R. 1994. Confidence intervals for regression quantiles. Pages 349–359 Asymptotic statistics. Springer.
Koenker, R. 2001. Quantile Regresssion. Encyclopedia of Environmetrics.
Kuo, F. E., and W. C. Sullivan. 2001. Environment and Crime in the Inner City: Does Vegetation Reduce Crime? Environment and Behavior 33:343–367.
Leitzinger, C. 2001. 4.2 The Potential of Co-composting in Kumasi–Quantification of the Urban and Peri-urban Nutrient Balance. Waste Composting for Urban and Peri-urban Agriculture: Closing the Rural-Urban Nutrient Cycle in Sub-Saharan Africa:150.
Linden Hills Power and Light. 2014. About Us - LHPower&Light. http://www.lhpowerandlight.org/about-us.html.
Lindon, M., and S. Heiskary. 2009. Blue-green algal toxin (microcystin) levels in Minnesota lakes. Lake and Reservoir Management 25:240–252.
Liu, C., C. Zou, Q. Wang, Y. Hayashi, and T. Yasunari. 2014. Impact assessment of human diet changes with rapid urbanization on regional nitrogen and phosphorus flows—a case study of the megacity Shanghai. Environmental Science and Pollution Research 21:1905–1914.
Loper, S. J., A. L. Shober, C. Wiese, G. C. Denny, and C. D. Stanley. 2013. Nutrient Leaching during Establishment of Simulated Residential Landscapes. Journal of Environment Quality 42:260.
Lovett, G. M., and C. L. Goodale. 2011. A New Conceptual Model of Nitrogen Saturation Based on Experimental Nitrogen Addition to an Oak Forest. Ecosystems 14:615–631.
Lovett, G. M., K. C. Weathers, and M. A. Arthur. 2002. Control of Nitrogen Loss from Forested Watersheds by Soil Carbon:Nitrogen Ratio and Tree Species Composition. Ecosystems 5:712–718.
Martini, N. F., and K. C. Nelson. in press. The role of knowledge in residential land management. Urban Ecosystems.
Martini, N. F., K. C. Nelson, and M. E. Dahmus. in press. Exploring neighborhood niffusion of yard management knowledge as a step toward Improved urban ecosystems. Environmental Management.
Martini, N. F., K. C. Nelson, S. E. Hobbie, and L. A. Baker. 2013. Why "Feed the Lawn"? Exploring the Influences on Residential Turf Grass Fertilization in the Minneapolis-Saint Paul Metropolitan Area. Environment and Behavior.
McCarron, J. K., A. K. Knapp, and J. M. Blair. 2003. Soil C and N responses to woody plant expansion in a mesic grassland. Plant and Soil 257:183–192.
McPhearson, T., R. Auch, and M. Alberti. 2013. Regional Assessment of North America: Urbanization Trends, Biodiversity Patterns, and Ecosystem Services. Pages 279–286 Urbanization, Biodiversity and Ecosystem Services: Challenges and Opportunities. Springer.

McPherson, E. G., J. R. Simpson, Q. Xiao, and C. Wu. 2011. Million trees Los Angeles canopy cover and benefit assessment. Landscape and Urban Planning 99:40–50.
Meinzinger, F., K. Kröger, and R. Otterpohl. 2009. Material flow analysis as a tool for sustainable sanitation planning in developing countries: case study of Arba Minch, Ethiopia. Water Science & Technology 59:1911.
Metropolitan Council. 2014. Environmental Services. http://www.metrocouncil.org/About-Us/Facts/Wastewater-WaterF/FACTS-Environmental-Services.aspx.
Milesi, C., S. W. Running, C. D. Elvidge, J. B. Dietz, B. T. Tuttle, and R. R. Nemani. 2005. Mapping and Modeling the Biogeochemical Cycling of Turf Grasses in the United States. Environmental Management 36:426–438.
Miller, R. W. 1997. Urban forestry: planning and managing urban greenspaces. Second edition. Prentice Hall.
Minnesota Energy Assistance Program. 2013. EAP Policy Manual FFY2014.
Minnesota Pollution Control Agency. 2012. 2012 Minnesota water quality: Surface water section. Report to the Congress of the United States, water years 2010-2011.
Minnesota Pollution Control Agency. 2013a. Minnesota - Lake Integrated Assessments for Upper Mississippi River Basin (Upper Portion), based on the 2012 MN Integrated Report to Congress of the United States.
Minnesota Pollution Control Agency. 2013b. Air quality in Minnesota: 2013 report to the Legislature.
Morée, A. L., A. H. W. Beusen, A. F. Bouwman, and W. J. Willems. 2013. Exploring global nitrogen and phosphorus flows in urban wastes during the twentieth century. Global Biogeochemical Cycles 27:836–846.
Munger, J. W., S. C. Wofsy, P. S. Bakwin, S.-M. Fan, M. L. Goulden, B. C. Daube, A. H. Goldstein, K. E. Moore, and D. R. Fitzjarrald. 1996. Atmospheric deposition of reactive nitrogen oxides and ozone in a temperate deciduous forest and a subarctic woodland: 1. Measurements and mechanisms. Journal of Geophysical Research: Atmospheres 101:12639–12657.
Nassauer, J. I., and P. Opdam. 2008. Design in science: extending the landscape ecology paradigm. Landscape Ecology 23:633–644.
Nelson, K. C., S. Grayzeck, J. Y. King, S. E. Hobbie, L. A. Baker, and J. P. McFadden. 2008. Our household choices in urban living survey. University of Minnesota, Saint Paul.
Nowak, D. J. 1996. Notes: Estimating Leaf Area and Leaf Biomass of Open-Grown Deciduous Urban Trees. Forest Science 42:504–507.
Nowak, D., D. Crane, R. Hoehn, J. Stevens, and J. Walton. 2006a. Assessing Urban Forest Effects and Values: Minneapolis' Urban Forest. US Forest Service, Northeastern Research Station.
Nowak, D. J., D. E. Crane, and J. C. Stevens. 2006b. Air pollution removal by urban trees and shrubs in the United States. Urban Forestry & Urban Greening 4:115–123.
Office of Energy Efficiency and Renewable Energy. 2011. Emission factors for fuel combustion from natural gas, LPG, and oil-fired residential appliances. United States Department of Energy.
Parfitt, J., M. Barthel, and S. Macnaughton. 2010. Food waste within food supply chains: quantification and potential for change to 2050. Philosophical Transactions of the Royal Society B: Biological Sciences 365:3065–3081.

Peters, E. B., and J. P. McFadden. 2010. Influence of seasonality and vegetation type on suburban microclimates. Urban Ecosystems 13:443–460.
Peters, E. B., J. P. McFadden, and R. A. Montgomery. 2010. Biological and environmental controls on tree transpiration in a suburban landscape. Journal of Geophysical Research: Biogeosciences 115.
Peters, E. B., R. V. Hiller, and J. P. McFadden. 2011. Seasonal contributions of vegetation types to suburban evapotranspiration. Journal of Geophysical Research: Biogeosciences 116:n/a–n/a.
Pickett, S. T. A., M. L. Cadenasso, J. M. Grove, C. G. Boone, P. M. Groffman, E. Irwin, S. S. Kaushal, V. Marshall, B. P. McGrath, and C. H. Nilon. 2011. Urban ecological systems: Scientific foundations and a decade of progress. Journal of Environmental Management 92:331–362.
Qian, X. (Lisa), K. C. Nelson, L. A. Baker, S. E. Hobbie, J. Y. King, and J. P. McFadden. in review. C flux from household natural gas and electricity use: Assessing the effects of demographic factors, house characteristics, house thermal properties, and human behaviors.
Qin, Z., A. L. Shober, R. C. Beeson, and C. Wiese. 2013. Nutrient Leaching from Mixed-Species Florida Residential Landscapes. Journal of Environment Quality 42:1534.
Rabalais, N. N., R. E. Turner, and W. J. W. Jr. 2002. Gulf of Mexico Hypoxia, a.k.a. "The Dead Zone". Annual Review of Ecology and Systematics 33:235–263.
Rabalais, N. N., R. J. Díaz, L. A. Levin, R. E. Turner, D. Gilbert, and J. Zhang. 2010. Dynamics and distribution of natural and human-caused hypoxia. Biogeosciences 7:585–619.
Ramírez, C. A., and E. Worrell. 2006. Feeding fossil fuels to the soil. Resources, Conservation and Recycling 46:75–93.
Ramsey County. 2012. Beyond the garbage can: Ramsey County's solid waste master plan 2011-2030.
Rhoades, C. 1997. Single-tree influences on soil properties in agroforestry: lessons from natural forest and savanna ecosystems. Agroforestry Systems 35:71–94.
Saint Paul Department of Parks and Recreation. 2010. Street and park tree master plan, City of Saint Paul, Minnesota.
Saint Paul Department of Parks and Recreation. 2013. Park and boulevard tree planting list.
Selbig, W. R., and R. T. Bannerman. 2007. Evaluation of Street Sweeping as a Stormwater-Quality-Management Tool in Three Residential Basins in Madison, Wisconsin, Scientific Investigations Report 2007-5156. US Geological Survey, Reston, Virginia.
Siegrist, R. L., M. D. Witt, and W. C. Boyle. 1976. The characteristics of rural household wastewater. Small Scale Waste Management Project, University of Wisconsin–Madison and University of Wisconsin–Extension.
Smith, V. H. 1998. Cultural Eutrophication of Inland, Estuarine, and Coastal Waters. Pages 7–49 *in* M. L. Pace and P. M. Groffman, editors. Successes, Limitations, and Frontiers in Ecosystem Science. Springer New York.
Solid Waste Management Coordinating Board. 2007. Source separated food waste and organics management report. Page 84.
Sorenson, J. R. 2013. Potential Reductions of Street Solids and Phosphorus in Urban Watersheds from Street Cleaning, Cambridge, Massachusetts, 2009-2011. US

Department of the Interior, US Geological Survey.
Trojan, M. D., J. S. Maloney, J. M. Stockinger, E. P. Eid, and M. J. Lahtinen. 2003. Effects of land use on ground water quality in the Anoka Sand Plain Aquifer of Minnesota. Ground Water 41:482–492.
United Nations Population Fund. 2007. State of world population 2007: Unleashing the potential of urban growth. United Nations Population Fund, New York.
University of Minnesota Extension. 2008. Composting and Mulching: A Guide to Managing Organic Yard Wastes. http://www.extension.umn.edu/distribution/horticulture/components/3296-04.html.
US Energy Information Authority. 2002. Updated state-level greenhouse gas emission coefficients for electricity generation. 1998–2000. Energy Information Administration, U.S. Department of Energy.
US Energy Information Authority. 2011, January 31. Voluntary Reporting of Greenhouse Gases Program - Fuel Emission Factors. http://www.eia.gov/oiaf/1605/coefficients.html.
US Energy Information Authority. 2013, February. State historical tables for 2011.
US Environmental Protection Agency. 1994. Guide to Septage Treatment and Disposal.
US Environmental Protection Agency. 1995. Technology Transfer Network Clearinghouse for Inventories and Emission Factors. Fifth Editions. Compilation of air pollutant emission factors, vol. 1. Stationary points and area sources.
US Environmental Protection Agency. 2007. Nitrates and Nitrites: TEACH Chemical Summary.
USDA, N. 2014. The PLANTS database. plants.usda.gov.
Vetaas, O. 1992. Micro-site effects of trees and shrubs in dry savannas. JOURNAL OF VEGETATION SCIENCE 3:337–344.
Vinneras, B. 2002. Possibilities for sustainable nutrient recycling by faecal separation combined with urine. Swedish University of Agricultural Sciences, Uppsala, Sweden.
Walker, R. V., M. B. Beck, and J. W. Hall. 2012. Water — and nutrient and energy — systems in urbanizing watersheds. Frontiers of Environmental Science & Engineering 6:596–611.
Waschbush, R. J., W. R. Selbig, and R. T. Bannerman. 1999. Sources of Phosphorus in Stormwater and Street Dirt from Two Urban Residential Basins In Madison, Wisconsin, 1994–95. Page 51. US Geological Survey, Middleton, WI.
Weirich, C. A., and T. R. Miller. 2014. Freshwater Harmful Algal Blooms: Toxins and Children's Health. Current Problems in Pediatric and Adolescent Health Care 44:2–24.
Wright, I. J., P. B. Reich, M. Westoby, D. D. Ackerly, Z. Baruch, F. Bongers, J. Cavender-Bares, T. Chapin, J. H. C. Cornelissen, M. Diemer, J. Flexas, E. Garnier, P. K. Groom, J. Gulias, K. Hikosaka, B. B. Lamont, T. Lee, W. Lee, C. Lusk, J. J. Midgley, M.-L. Navas, Ü. Niinemets, J. Oleksyn, N. Osada, H. Poorter, P. Poot, L. Prior, V. I. Pyankov, C. Roumet, S. C. Thomas, M. G. Tjoelker, E. J. Veneklaas, and R. Villar. 2004. The worldwide leaf economics spectrum. Nature 428:821–827.
Xiao, Q., E. G. McPherson, J. R. Simpson, and S. L. Ustin. 1998. Rainfall interception by Sacramento's urban forest. Journal of Arboriculture 24:235–244.

Ye, S. 2011. Minnesota Crops Profile. Minnesota Department of Agriculture.